Reconciling theory and practice in environmental accounting

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RECONCILING THEORY AND PRACTICE IN ENVIRONMENTAL ACCOUNTING

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door

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1.

Introduction

1.1 Motivation and relevance for policy

Economies crucially depend on the environment both for provisioning of inputs of natural resources and ecosystem services as well as for functioning as a sink for its emissions and waste. A concern with standard national accounting (as reflected in the System of National Accounts (SNA); UN et al., 2009) is that it does not properly take this dependency of economic activity on the environment into account. For instance, suppose a country were to cut its entire forested area for harvesting timber within a single year. According to standard accounting conventions, the value of production and income that is derived would be included in full in macroeconomic indicators such as net domestic product as no cost of depleting natural capital is taken into consideration. This is problematic as it results in an inconsistent treatment of fixed capital and natural capital as the former is depreciated when used up in production but the latter is not. More importantly, there is a sustainability issue involved: when the depletion of resources hampers future production possibilities, indicators such as net national income when unadjusted for the cost of depletion yield incorrect signals to policy makers about future welfare. The treatment of externalities may serve as a second example. Externalities – either positive in the form of the supply of ecosystem services or negative in case of environmental degradation – are not recorded in national accounts as they do not constitute 'transactions' i.e. actions undertaken by mutual agreement between two institutional units. There are however growing concerns about the state of the environment (IPCC, 2007; Rockström et al., 2009) as a result of environmental degradation due to pollution, which is therefore not reflected in standard indicators used to monitor economic activity.

These issues have been studied for a long time by statisticians (in particular national accountants) and (environmental) economists alike. It is useful at this stage to distinguish between *green accounting* (the notion used most frequently in the research community/theoretical literature) and *environmental accounting* – often called environmental-economic accounting (the notion used predominantly in the statistical community/empirical literature). While both fields have a shared ambition to develop better measures of progress that take environmental concerns into account, in terms of the key research questions as well as methodology there exist stark differences. The green accounting literature has traditionally focused on studying the relationship between the concepts of welfare, income and wealth, in the setting of theoretical models which include issues such as extraction of natural resources, pollution and treatment of ecosystem services (e.g. Dasgupta and Heal, 1974; Weitzman, 1976; Hamilton, 1996; Arrow et al., 2003a; Asheim and Wei, 2009; Dasgupta, 2009; Barbier, 2013). By contrast, the environmental accounting community has followed a more pragmatic approach, focusing on how to integrate the use of environmental assets into the national accounts. It is important to clarify, that although environmental accounting may mean different things in various contexts, we will define it in this thesis in a narrow sense as so-called satellite accounts of the SNA as described in the System of Environmental-Economic Accounting (SEEA, UN et al., 2012; 2013). As will be explained in greater detail in Chapter 2, satellite accounts respect the core definitions and classifications that underlie the SNA, but at the same time allow for flexibility, such as an extended asset or production boundary or additional classifications by purpose (Edens and De Haan, 2010).

Unfortunately, as noted by Heal and Kriström (2005, p.1151) in their extensive review of green accounting, a gap exists between the theoretical and empirical literature: *"there is a gap between theory and practice: empirical studies are not always backed up by sound theory"*. There are several explanations for this dichotomy. First of all, at a rudimentary level there appears to be disagreement about whether accounting frameworks such as the SNA or the SEEA require a theoretical economic foundation at all. To give an example, although the development of national accounts is closely tied to the advent of Keynesian macro economics (Vanoli, 2005, p.19 refers to it as the "skeleton of National Accounting"), recent versions of the SNA (the 1993 SNA, UN et al., 1993; the 2008 SNA, UN et al., 2009) rather claim to be applicable to any school of thought.[1] In the same vein, the SEEA 2003 (UN et al., 2003) describes three approaches to sustainable development (the three-pillar approach; the ecological approach; and the capital approach) but emphasizes that it does not favor one approach over the other: *"the system has not been designed to serve any particular perspective and, indeed, should be of considerable value regardless of the user's particular point of view on the concept."(UN et al., 2003, para. 1.34)*[2] In short, accounting frameworks increasingly aspire to be 'theory-neutral'. At the same time (environmental) accounting practices – especially their use of environmentally adjusted aggregates – are often criticized for failing to explain how they are related to economic theory e.g.: "*we take the view that the SEEA ... has yet to provide a clear answer about what the system is supposed to measure. It is not a Keynesian style set of accounts for macro-economic*

[1] "The types of macroeconomic models ... vary according to the school of economic thought of the investigator as well as the objectives of the analysis, but the SNA is sufficiently flexible to accommodate the requirements of different economic theories or models, provided only that they accept the basic concepts of production, consumption, income, etc. on which the SNA is based." (2008 SNA, p.5)

[2] Of course there could also be political and practical reasons in terms of acceptability and creating ownership that may underlie this stance.

purposes, neither does it provide welfare – indices and, as far as we can tell, it does not necessarily produce a measure of sustainable income" (Heal and Kriström, 2005).[3)] It is also exemplary that while the SEEA 2003 mentioned Hicksian income several times, the SEEA Central Framework (SEEA CF, UN et al., 2012) is devoid of any references to theoretical contributions on green accounting.

On the other hand, from the perspective of statisticians theoretical work on green accounting often neglects data availability issues for instance when assumptions are made about the economy being on an optimal path or when theories are built upon unobservable 'shadow prices'. By contrast, statisticians often depart from a stock taking of available data sources. A third issue is more practical: environmental accountants often do not publish in academic journals and their work is therefore often not well-accessible for the academic community, while the theoretical work is often difficult to understand as the techniques involved (optimal control, Hamiltonians etc..) are not part of the standard toolbox of statisticians.

As a result, there appears to be little cross-fertilization and plenty of misunderstandings between theory and practice.[4)] This is unfortunate as there is a growing recognition of the importance of green/environmental accounting in a number of domains. For instance, the need to complement traditional indicators such as gross domestic product (GDP) with indicators that take environmental concerns into account has been emphasized within the GDP and Beyond Roadmap (European Commission, 2009), by the Stiglitz-Sen-Fitoussi report (Stiglitz et al., 2009), and, recently, in the Rio+20 declaration of the United Nations Conference on Sustainable Development "The future we want" (UN, 2012) which emphasized in the much contested paragraph 38 "the need for broader measures of progress to complement gross domestic product".

Secondly, there is a growing interest in better understanding the economic implications of the ongoing changes to the world's ecosystems (MA, 2005; TEEB 2010; EC 2011; UK NEA, 2011), as evidenced by the Nagoya Protocol of the Convention on Biological Diversity. This is reflected in, for example, the recent EU Biodiversity strategy (EC, 2011) which calls upon Member States to *"assess the state of ecosystems and their services in their national territory by 2014 and assess the economic value of such services, and promote the integration of these values into accounting*

3) Another example: "The (green accounting) literature clearly provides the answer to one important question: What is the change in total asset value when a resource is extracted? But it does not answer another question which should be profoundly important to policy makers: How much has social welfare changed when this resource is extracted?" (Hamilton and Ruta, 2009)

4) For instance Heal and Kriström (2005, p.1204) credit Ahlroth (2001) as "one of the few attempts that now exists to bridge a gap between two traditions in the green accounting literature."

and reporting systems at EU and national level by 2020". This requires however bridging practices from ecological economics and environmental economics (in particular valuation studies) with accounting conventions, an emerging area which is called *ecosystem accounting*. Also the World Bank Group's 50:50 campaign launched in the wake of Rio+20 and the World Bank's WAVES (Wealth Accounting and the Valuation of Ecosystem Services) partnership underline the policy interest in better accounting for natural capital.

The motivation behind this thesis is to reconcile theory and practice in the area of green/environmental accounting.

1.2 Objective and structure of the thesis

The main research question of this thesis is the following:

What are the possibilities to narrow the gap between theory and practice in green / environmental accounting?

Practice is understood here as the activities carried out by practitioners of environmental accounting (by and large) situated within the statistical community. Theory is understood here – in a broad sense – as the activities carried out by those situated (by and large) outside the statistical community such as researchers working in areas such as green accounting, ecosystem services or input-output analysis. The distinction is therefore primarily made at the level of communities that can be characterized by the use of different conventions and principles, as reflected in handbooks, study books, and articles – respectively referred to as the empirical and theoretical literature.[5]

Underneath this question lie two sub questions, in particular:

What are the main causes for the existence of a gap between theory and practice in green/environmental accounting?

[5] Therefore, theoretical literature is used here not in a narrow sense as articles restricted to models without empirical corroboration, but for articles whose point of departure is the formulation of a theory or model, whereas the point of departure in the empirical literature is often observation.

Would it be possible to strengthen environmental accounting practices by underpinning them with a theoretical foundation?

The theoretical and empirical literature on green/environmental accounting has become very extensive, especially in recent years. Therefore, in terms of scope, this thesis will have a clear focus on valuation issues. This is also the area in which a lot of challenges still lie ahead. There is also a strong tradition in environmental accounting that focuses on measuring physical flows, but this area has reached international standardization with the acceptance of the SEEA Central Framework (SEEA CF) as a statistical standard. The main approach chosen in this thesis is to address the research questions by investigating how theory and practice relate in different areas of study: the extraction of a non-renewable natural resource which addresses the issue of how to cost depletion; the emerging area of ecosystem accounting in which issues are investigated such as how to cost degradation as well as how to integrate ecosystem services into an accounting framework; wealth accounting; and finally, applications of environmental accounting in the form of environmentally extended input-output analysis. This results in the following outline of the thesis.

First, as not all environmental economists may be aware of the specificities of environmental accounting practices, Chapter 2 provides some necessary context: a brief history of environmental accounting; a description of environmental accounting in practice; an introduction of the SEEA conceptual framework. It will also provide an overview of country practices in environmental accounting that are relevant from a valuation perspective such as wealth accounting and experiences with the compilation of 'green GDP'.

Chapter 3 reviews a number of theoretical and empirical approaches towards estimating costs of depletion that have been recently brought forward in the context of environmental accounting and green accounting: depletion as the change in total wealth (UN et al., 2003); depletion as 'using up' of the resource as proposed in the SEEA CF (UN et al., 2012; depletion as 'net saving' (World Bank, 2011); or, depletion as net investment (Asheim and Wei, 2009). The differences in assumptions between these measures are clarified by contrasting their approaches with the classic theory of a firm engaged in extraction. All measures are evaluated using a time series of data on Dutch natural gas reserves.

In Chapter 4 we push the frontier of environmental accounting further by investigating the emerging topic of *ecosystem accounting* building on the process that led to the SEEA Experimental Ecosystem Accounting (SEEA EEA; UN et al., 2013). Here we compare the basic concepts of environmental and national accounts with those used in ecological and environmental economics. Although there is a growing understanding of and availability of data on ecosystem services there is still very limited experience with the integration of such values into national accounts. This chapter identifies four key methodological challenges in developing ecosystem accounts: the definition of ecosystem services in the context of accounting; their allocation to institutional sectors; the treatment of degradation and rehabilitation; and, valuing ecosystem services consistent with SNA principles. The different perspectives taken on these challenges are analyzed and a number of proposals to address these challenges are presented.

The change in a country's national wealth over time provides an indication to what extent its development is being sustainable (World Bank, 2011; UNU-IHDP and UNEP, 2012). While the initial focus in this area was on assessing changes in wealth through (net) investments, which resulted in well-known indicators of sustainability such as genuine savings (or adjusted net savings (ANS); Hamilton 1996) or policy prescriptions for achieving sustainability such as Hartwick's rule (Hartwick, 1977), in recent contributions (World Bank, 2006, 2011; UNU-IHDP and UNEP, 2012) the objectives have broadened towards obtaining estimates of the various capital assets that are the constituents of wealth, an area which we refer to as *wealth accounting*. The World Bank (2011) recently published times series of comprehensive wealth and adjusted net savings (ANS) estimates for over 120 countries. Chapter 5 reviews and where possible refines these estimates for the Netherlands, by comparing them with official Dutch statistics. The chapter also contains a critical review of Mirko and Ferreira (2011) that performed a similar exercise for Ireland. The chapter provides a number of suggestions for future directions of research in order to improve wealth accounting.

Chapter 6 contains an application of environmental accounts based upon environmentally extended input-output analysis. These techniques allow calculating the emissions embodied in the consumption of goods and services of countries, which may subsequently be expressed in the form of a carbon footprint or an emission trade balance(s). We construct bilateral emission trade balances for the Netherlands with 17 countries/regions and compare results for 1996 and 2007 for three different greenhouse gases. We establish a cross-sectional analysis of bilateral emission trade balances into a volume of trade, composition and technology effect. In order to analyze the driving forces of changes over time, we perform a structural decomposition analysis of embodied import and export emissions. The chapter fits into this thesis as it compares practices from the input output community with practices in the statistical community. In particular, we investigate data discrepancies between international data sources and official statistics.

Finally, Chapter 7 returns to the main research questions and provides a summary of the main outcomes of this thesis. Herewith, this thesis hopes to contribute to a better understanding of the issues around recording and valuation of natural capital (in particular depletion; ecosystem services; wealth) in accounting frameworks, in order to improve the information basis for policy makers.

2. Overview of environmental accounting

2.1 History[1)]

The origins of environmental accounting can be traced to the (late) 1970s, when several European countries initiated work independently of each other (Hecht 2005, p.9). In 1978, the Norwegian Environment Ministry commissioned Statistics Norway to develop natural resource accounts as a tool to better manage natural resources and the environment (Alfsen, 1996, p.5). This was due to growing environmental concerns because of intensive expansion of hydropower, over-exploitation of fish stocks and the discovery of significant oil and gas reserves. Also Denmark was an early adaptor, it started the compilation of energy flow accounts around 1975. This was triggered by the 1973 oil crisis that generated a lot of interest in energy issues, especially in energy saving and improving energy efficiency. In the 1980s, France developed an accounting system to assess, both quantitatively and qualitatively, the state and evolution of its 'natural patrimony' (Vanoli, 2005, p.344). These initial efforts have in common that they were focused on obtaining physical descriptions of natural resource use.

Regarding monetary descriptions, Vanoli (2005, p.281) in his History of National Accounting credits Kuznets for initiating *"a long tradition ... aimed at making national income ... an indicator of economic welfare"* by means of all sorts of adjustments to conventional GDP. Nordhaus and Tobin (1972) is a highly influential study that estimates macro-economic welfare for the United States, but their focus was not on the environment.[2)] Peskin (1976) is a theoretical contribution which discusses in general terms how values of service flows and damages associated with the use of environmental assets can be integrated into the national accounts. However, according to Vanoli (ibid., p.294), it was not until the second part of the 1980s that the focus was really placed on adjusting indicators such as GDP and NDP for the use of natural capital, rather than obtaining asset values. This development was influenced by growing concerns that these indicators did not properly take the depletion and degradation of natural assets into account as a result of economic activity. In the Netherlands, the economist Hueting was influential in his ambitious efforts to estimate a sustainable national income taking depletion and degradation

1) Sections 2.1, 2.2 and 2.3 of this chapter are based on Edens (2012b) and Edens and De Haan (2010); Section 2.3 also draws upon Edens and Hein (2013); Section 2.4 is based on Chapter 8 of World Bank (2011) for which the author did the research and wrote the initial draft in a capacity as consultant; some material in this Chapter has been drawn from a study commissioned by the World Bank on Lessons Learned from Environmental Accounting for which about 20 persons either currently in charge of environmental accounting programs or with detailed knowledge of environmental accounting were performed (2012; not published, although some of the findings were disseminated through World Bank (2012)).

2) Interestingly, Nordhaus and Tobin (1972) state: "If we had estimates of the value of environmental capital, we could add them to the national wealth estimates ... and modify our calculations of MEW [measure of economic welfare] net investment accordingly. We have not been able to make this, adjustment, but given the size of the other components of wealth, we do not believe it would be significant."

of the environment into account (Hueting, 1980). This later triggered the development of physical flow accounts or NAMEA (National Accounting Matrix including Environmental Accounts), that present physical information alongside economic information to allow them to be comparable (De Haan and Keuning, 1996; De Haan, 2004) – see Box 2.1.

In developing regions, during the 1980-ies, the World Bank and the United Nations Environmental Programme (UNEP) sponsored several workshops (see Ahmad et al., 1989), which led to the conclusion that 'enough progress had been made to link environmental accounting to the ... SNA' (Lutz, 1993). In response, the United Nations Statistics Division (UNSD – at the time referred to as UNSTAT) started working on a framework that was discussed at several sessions of the International Association for the Review of Income and Wealth (IARIW). A highly influential study was undertaken by the World Resources Institute (Repetto et al., 1989) which estimated the depreciation costs of Indonesia's natural resources and showed that this would lead to a significant downward adjustment of its growth rates. Between 1989 and 1992, the World Bank and UNSD conducted several pilot country studies (e.g. Mexico, Papua New Guinea), in order to test the accounting framework that was under development, which was eventually published by the United Nations as Handbook of National Accounting: Integrated Environmental and Economic Accounting (1993 SEEA: UN, 1993).

Most of the environmental accounting programs – in both the developed and developing world – however were initiated in the early 1990-ies. The Brundtland report (UN, 1987) that came out in 1987 sparked a lot of interest in sustainable development. The "Earth Summit" held in 1992 in Rio de Janeiro was a major stimulus for environmental accounting as it called in its Agenda 21 (UN, 1992) for *"establishing systems for integrated environmental and economic accounting ... in all member States at the earliest date, with the main objective to expand existing systems of national economic accounts in order to integrate environment and social dimensions in the accounting framework"*. As the preface of the 1993 SEEA clearly states, the handbook was work in progress, and there was a clear need to continue conceptual discussions. To this end the statistical community established the London Group on Environmental Accounting, a forum for expert practitioners, from the increasing number of countries (both developed and developing) that had started environmental accounting programs. In 2000, Integrated Environmental and Economic Accounting – An Operational Manual (UN, 2000), was published by UNSD and UNEP, followed in 2003 by the SEEA 2003 (UN et al., 2003). Although the SEEA 2003 was a major step forward, it still did not provide unique recommendations on a number of issues. Therefore, the UN Statistical Commission established the United Nations Committee of Experts on Environmental-Economic Accounting

(UNCEEA) at its 36th Session in March 2005 with as one of its main objectives starting a process of developing the SEEA into an international statistical standard.

Table 2.1 Timeline with milestones in environmental accounting

1970-ies	Norway and Denmark embark upon environmental accounting
1980-ies	Netherlands and France embark upon environmental accounting
1983-1988	World Bank and UNEP sponsored workshops
1989	World Resources Institute study on Indonesia (Repetto et al., 1989)
1989	Framework on integrated environmental accounting presented at 21st session of IARIW
1989-1992	World Bank/UNSTAT pilot projects in Mexico and Papua New Guinea
1991	IARIW special conference on environmental accounting (Baden, Austria)
1992	'Earth Summit' in Rio; Agenda 21 Ch. 8 recognizes environmental accounting
1993	SEEA 1993 published (UN, 1993)
1994	London Group on Environmental Accounting established
1994	EU green accounting strategy adopted
1994	US environmental accounting activities stopped due to political opposition
1995	Eurostat and Statistics Netherlands jointly organize a workshop on the NAMEA concept
1997	Executive Order signed which institutionalizes the Philippine Economic Environmental and Natural Resources Accounting
2000	SEEA Operational Manual published
2000	Mexican General Law of Ecological Equilibrium and Environmental Protection requests compilation of Ecological Net Domestic Product
2003	SEEA-2003 published as white paper
2003	EU Strategy for Environmental Accounting endorsed
2005	UNCEEA established by UN Statistical Commission to revise SEEA 2003
2006	China publishes 'green GDP' figures
2008	Revised EU environmental accounting strategy
2011	EU legal base for environmental accounting adopted
2011	Dutch report "Green growth in the Netherlands" commended by OECD SG
2011	Expert Group on greening India's National Accounts established
2012	SEEA Central Framework adopted by UN Statistical Commission as international statistical standard
2013	SEEA Experimental Ecosystem Accounting brought to UN Statistical Commission

Source: Author's characterization.

Early on in the revision process, it was decided that the revised SEEA would consist of three separate volumes. Volume 1 was to focus on those accounts that were considered mature enough to be included in a statistical standard; volume 2 would consists of those topics for which consensus could not be reached or for which country experiences are limited but which are expected to be highly policy relevant; and volume 3 consists of the applications of the accounts presented in volumes 1 and 2. Volume 1 which later came to be known as the SEEA Central Framework (SEEA CF, UN et al., 2012) was adopted as an international statistical standard by the UN Statistical Commission in 2012. Volume 2 – entiled SEEA Experimental Ecosystem Accounting (SEEA EEA, UN et al., 2013) – and volume 3 – entitled SEEA Extensions and Applications were discussed during the

44th session of the UN Statistical Commission in early 2013. The SEEA EEA is not recognized as a statistical standard but as state of the art conceptual framework that can be used by countries for further testing and experimentation.

Box 2.2 Environmental accounting in the Netherlands

Dutch interest in environmental accounting was pioneered by the efforts of Hueting in the late seventies to measure sustainable national income defined as the maximum level of production that is attainable while vital environmental functions remain intact (Hueting, 1980). This led to fierce discussions and was considered by some to lie outside the realm of statistics as it required modeling (Hecht, 2005). It inspired however the development of a so-called NAMEA (National Accounting Matrix including Environmental Accounts) around 1990. A NAMEA is a matrix presentation of the sequence of accounts extended with an 'environmental module' (De Boo et al., 1991).

The main motivations for the NAMEA were to go beyond monetizing environmental flows, by introducing physical information alongside economic information to allow them to be comparable (De Haan et al., 1993), and to look at the whole sequence of accounts and not only at production. With its focus on describing physical flows and providing a disaggregation by purpose of transactions already covered in the National accounts, the NAMEA approach had the advantage that it remains within the purview of standard statistical practice. Another feature of the NAMEA presentation is that it presents information according to several environmental themes (such as 'acidification') which enables linking information to formulated Dutch policy objectives (De Haan and Keuning, 1996; De Haan, 2004).

While the initial focus of the NAMEA was on air emissions and energy use, several pilot studies followed on environmental protection expenditure, land use by industry, and on water accounts. The program during all these years practically consisted of about one person. This changed around 2005 when due to concurrent requests the program started to expand. First, the work of Eurostat in the area of environmental accounting and the ongoing legislation at European level, have been very important for the development and implementation of several key accounts. Second, specific policy demands at the national level have led to the development of a number of specific accounts that are now published on a regular basis. Examples are the water accounts, which serve the data requirements for reporting to the Water Framework Directive, and the Economic Radar of the sustainable energy sector commissioned by the Ministry of Economic Affairs. Finally, a further impetus was provided by a large statistical research program on measuring sustainable development and green

growth, which has also contributed to the development of several environmental accounts (emission permits, carbon footprint, etc.).
At present (2013) Statistics Netherlands has implemented a large part of the SEEA Central Framework. The focus has been on the compilation and dissemination of physical and hybrid flow accounts and monetary environmental activity accounts. Asset accounts have been less developed, primarily because the Netherlands has relatively few natural resources. The accounts are disseminated through an annual publication (Statistics Netherlands, 2009b, 2010b, 2011b, 2012b), via a dedicated webpage, in the electronic database of Statistics Netherlands (StatLine), and in various other publications of Statistics Netherlands. The NAMEA matrix itself however is no longer being disseminated. The accounts are also used for deriving indicators for measuring sustainable development as well as green growth (Statistics Netherlands, 2011c).

Within Europe, in 1994 a green accounting strategy was adopted and Eurostat started funding several pilot projects in countries. The European Strategy for Environmental Accounting was endorsed in 2003 followed by a revised version in 2008 by the Statistical Program Committee, which recommended (amongst others) to establish a legal base for environmental accounting. This was accomplished, when the Regulation on European Environmental Economic Accounts was adopted by the European Parliament and Council in July 2011 (EU, 2011).

2.2 Environmental accounting in practice

The SNA (UN et al., 2009) is an international statistical standard with specific guidelines on how to compile a set of interrelated accounts, which are designed to provide a comprehensive description of economic activity (e.g. production, consumption, and accumulation of assets). The SNA accomplishes this by describing the transactions (e.g. buying a product, or paying a tax) between so-called institutional units such as households or enterprises. These units can be classified either into institutional sectors (e.g. central government, or the financial sector) or into economic activities (colloquially called economic sectors) such as agriculture or mining.

Transactions are described in a sequence of accounts: the current accounts (production, distribution and use of income) provide information on production and value added by economic activities and various notions of income, with as main indicators gross domestic product (GDP), net national income (NNI), and savings. The accumulation accounts (capital, financial, other changes in volume) describe changes in assets by ownership. The resulting net worth and changes therein is recorded in the balance sheets.

The scope of the SNA is defined by a set of boundaries, most importantly the production boundary which defines when an activity is considered productive. For example, theft or cooking for household members is not considered a productive activity, but home growing of vegetables in kitchen gardens is included. Another important principle is that the national accounts are restricted to 'resident' institutional units, which is determined based on the economic territory of predominant economic interest. The national accounts include the activities of residents, regardless whether this activity occurs outside or inside a country's borders. For example, the production by someone who is temporarily sent abroad by his (resident) employer may be included. The accounts are therefore based upon economic considerations and do not follow citizenship, nationality, or mere geographical boundaries.

Satellite accounting was invented to allow for flexibility of the standard SNA conventions (Edens and De Haan, 2010). Well known examples are tourism satellite accounts and health accounts. The System for Environmental-Economic Accounts has been developed to provide a more comprehensive understanding of the interrelationship between economy and environment. The SEEA recognizes that economic activities critically depend on the environment both as a source of inputs such as natural resources, but also as a sink for its outputs in the form of emissions and waste. The SEEA integrates environmental statistics with economic statistics using the organizing principles (such as residence), classifications (such as the International Standard Industrial Classification of All Economic Activities – ISIC) and definitions of the SNA. At the same time, it takes a much broader perspective on the environment by expanding the SNA asset boundary. While the SNA defines assets in terms of two necessary conditions, namely that they should provide benefits and that they are owned, the SEEA relaxes both conditions and defines environmental assets more broadly as the naturally occurring living and non-living components of the Earth, together comprising the bio-physical environment, that may provide benefits to humanity (SEEA Central Framework, para 2.17). Another important aspect of the SEEA is that it complements the monetary scope of the SNA with physical descriptions of stocks and flows, for instance of stocks and changes over time of standing timber, quantities of water abstractions, and land cover accounts.

In the SEEA there is an explicit distinction of cultivated assets (e.g. a plantation) and natural assets (e.g. a natural forest). The SEEA also contains a set of accounts that describe environmental activities and transactions (e.g. taxes and subsidies) and environmental protection expenditure.

In practice, the work of an environmental accountant consists of integrating energy and environment statistics into economic statistics. Integration means adjusting data so that they match the concepts, definitions and classifications of the national accounts. For instance, while an energy balance provides a technical overview of the energy use and transformation that occurs within the geographical boundaries of a country, an energy account provides an economic picture of energy use by a country's resident units; it shows the supply and use of energy products by economic activities. Compiling an energy account, may therefore result in large adjustments due to a different treatment of international tourism and transport. Similarly, while emission inventories used for reporting on the Kyoto Protocol exclude greenhouse gas emissions inherent in international aviation in the national totals[3)], these emissions are included in air emission accounts in case the airline would be a resident unit.

The value added of environmental accounting is that the rigor of an accounting system due to its checks and balances (supply equals use; value divided by price should equal volume) may increase the reliability of data. It also ensures that all indicators derived from the accounting system are consistent with each other. On the other hand, critics of environmental accounting point out that having separate statistics such as an energy balance and an energy account, or Kyoto emissions and an air emission account, may confuse users.

2.3 Overview of SEEA

The SEEA CF consists of several types of accounts. The first category consists of physical flow accounts. They measure the use of the environment in terms of natural resource inputs and outputs of waste and emissions. Physical flow accounts can be expressed in different units resulting in energy accounts (in Joule), water accounts (in cubic meters), air emission accounts (in tons) and material flow accounts (in tons). Due to the use of common classifications and definitions, a one-to-one relationship of physical and monetary data is obtained, that allows compiling indica-

3) The emissions are reported as memorandum item.

tors on resource productivity or environmental efficiency disaggregated by industry. Physical flow accounts can be used to analyze the extent to which economic growth is being decoupled from resource inputs and pollution outputs. In order to achieve sustainability, absolute decoupling of environmental pressures from economic growth is considered a necessary condition by most scholars.

The second category of accounts in SEEA consists of monetary accounts that track environmentally related activities as well as policy instruments. Environmental protection expenditure accounts indicate how much a country spends on protecting or rehabilitating the environment. They also show what part of investments has an environmental purpose ('green investments'). Environmental goods and services accounts measure the size of activities related to the environment. Environmental tax accounts can be used to monitor whether a country's tax base is greening. Accounts for emission permits allow analyzing the incentives that different industries face to reduce their greenhouse gas emissions.

The third category of accounts in SEEA, are natural resource accounts that describe a country's natural capital (both renewable and non-renewable) in both physical and monetary terms. Analyzing time series of for instance stocks of timber, allows one to assess whether the natural capital base is being maintained over time in a sustainable manner. In monetary terms, natural resource accounts allow to derive extended measures of wealth such as compiled by the World Bank (2011). Natural resource accounts also allow estimating the value of depletion of various types of natural capital. This enables the calculation of 'green GDP' type of measures, which adjust common measures of production, income, and saving, towards more sustainable measures by correcting for depletion of natural capital.

While the SEEA CF provides a much broader perspective on the environment than the SNA, it does not provide an analysis of ecosystem services or ecosystem capital. One of the main reasons is that while the SEEA CF relaxes the asset boundary, it keeps the SNA production boundary intact. For produced assets such as a machine, the production boundary constrains the asset boundary, but this does not apply to many natural resources which are considered non-produced assets i.e. they are not the outcome of production processes and the services they provide are considered rent payments. Consequently, both the SNA and SEEA exclude from the production account various types of ecosystem services such as regulating services as well as the natural growth of biological assets. In addition, while the SEEA CF provides recommendations on the treatment of depletion, it does not contain a discussion of the treatment of environmental degradation or rehabilitation.

Although the SEEA Experimental Ecosystem Accounting has the same asset boundary as the SEEA CF, it provides a different perspective on natural capital (UN et al., 2013, para 1.15). Whereas the SEEA CF has a reductionist perspective conceiving of a forest as a collection of individual assets such as soil, timber, which provide the economy with market products (e.g. wood products), the SEEA EEA has a more holistic perspective conceiving of the forest rather as an ecosystem asset which provides a bundle of ecosystem services some of which are marketed (wood) some of which are not (carbon sequestration), hereby extending the SNA production boundary. The SEEA EEA proposes accounts in both physical and monetary units which describe the supply of ecosystem services as well as asset accounts for ecosystems. It also contains proposals for a carbon account and a biodiversity account.

2.4 Country practices

Environmental accounting programs

Figure 2.3 provides a rough indication of the current status of country experiences with environmental accounting. It cannot be emphasized enough that a lot of interpretation and judgment is involved in such a classification exercise and given that the field is undergoing rapid changes, the assessment is necessarily time-specific. That said, the overall picture that emerges is that at the point of assessment in 2012 about 72 countries – 33 developed and 39 developing – to date have experience with environmental accounting. The number of countries with a regular environmental accounting program – when at least one account is compiled regularly – is about 42, where we have included all EU member states. This is because the compilation of air emission, material flow and environmental tax accounts has become compulsory (by august 2013) for all EU member states (some derogations may apply to individual countries).[4] When we compare these results to earlier assessments – for instance Peskin and Lutz (1993) reviewed seven countries and mention that at least another eight countries are doing efforts, 15 in total – it is clear that environmental accounting is a growing area of statistics.

[4] To put these findings in context, the UNSD Global Assessment (UN, 2007) – based upon a written survey - found that 49 countries (out of 100 responding countries) have an environmental accounting program (29 developed and 20 developing). This larger number could be due to different interpretations of what having an environmental accounting program means as well as due to self-reporting. World Bank (2012), which contains a similar map, mentions that the number of countries with an existing program is 24 – although the map appears to show 28. This lower number may be due to an earlier time of assessment as well as a different interpretation of status for several countries.

Implementation of environmental accounting has been most extensive in developed countries. In Europe, there are about ten countries with long running environmental accounting programs. The focus in Europe has been primarily on physical flow accounts and monetary accounts for environmentally related activities and transactions. Outside Europe, Australia and Canada have had comprehensive programs since the early 1990s. The United States' environmental accounting activities however were stopped due to political opposition, shortly after its first publication in 1994. A panel was commissioned to review the work undertaken, which concluded that "*extending the U.S. national income and product accounts (NIPA) to include assets and production activities associated with natural resources and the environment is an important goal*" (Nordhaus and Kokkelenberg, 1999). The report however did not change the situation. In Asia, South Korea has a strong focus on wealth accounting, while Japan traditionally has a large interest in material flow accounting.

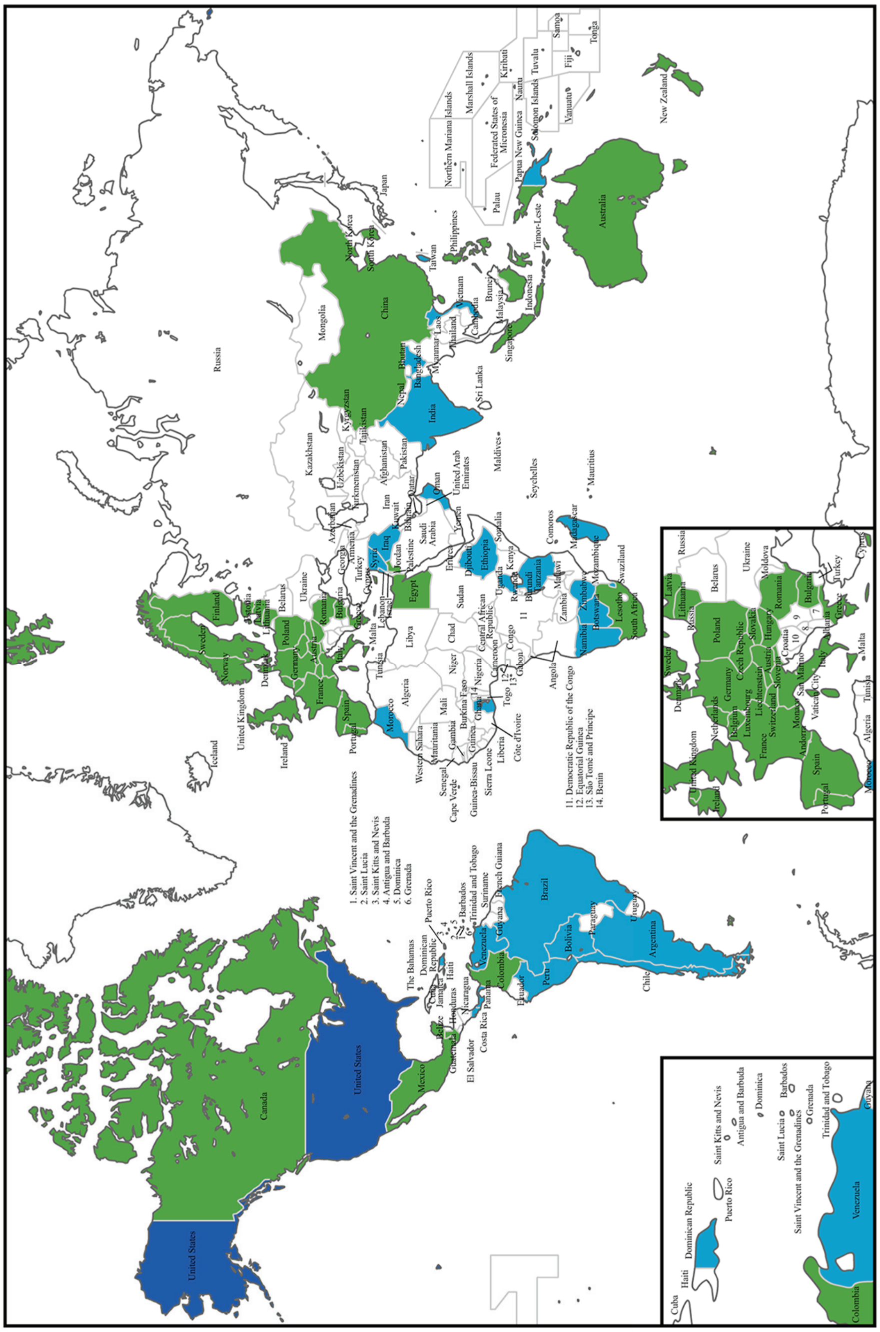
Several developing countries such as Mexico, South Africa, and Columbia have long running accounting programs. Some other developing countries have struggled to keep their programs running due to capacity constraints or lack of data. This situation is at the time of writing however changing rapidly due to the adoption of the SEEA CF as a statistical standard in 2012, which will be followed up by an implementation program. In parallel, there are international efforts such as The Economics of Ecosystems and Biodiversity (TEEB) and WAVES that are providing a large stimulus to environmental accounting in several developing countries.

As the focus in this thesis lies on valuation, we will focus now in greater detail on assessing country experiences in two domains: wealth accounting and assessments of 'green GDP'.

Country practices in wealth accounting

Although wealth is not an unambiguous term, in the context of wealth accounting it refers to what is sometimes called the capital (or stock based) approach, in which wealth is equated with 'the totality of resources which we are able to draw upon to support ourselves over time' (UNECE, 2009). Wealth is often broken down into different types of capital such as economic, natural, human, and social capital, although the precise asset boundary is under discussion (Hamilton, 2012). Another categorization of wealth frequently used is the asset classification of SNA that distinguishes between financial assets/liabilities and non-financial assets. Non-financial assets are broken down into produced assets (e.g. fixed assets such as dwellings and inventories) and non-produced assets (e.g. natural resources

Figure 2.3 Overview of country experiences with environmental accounting (2012; not exhaustive)[1)]



Multiple sources: UNSD, 2007; Hosn, 2011; Searchable Archive of Publications on Environmental-Economic Accounting (http://unstats.un.org/unsd/envaccounting/ceea/archive; websites of NSOs; personal interviews.
Legend: green = regular program; blue = pilot projects; dark blue = suspended.
1) Map was made using: http://edit.freemap.jp/en/trial_version/select_map/

or licences). Balance sheets provide an overview of all assets and liabilities of a country which results in an indicator called net worth. The definition and disaggregation of wealth will be further discussed in Chapter 5.

According to a 1979 survey of country practices in compiling balance sheets (Blades, 1980), only one country could compile a comprehensive balance sheet, seven countries compiled conventional balance sheets i.e. excluding for instance subsoil assets, and a further 31 countries were capable of providing some statistics on balance sheet items. In terms of non-produced assets, only two countries provided estimates of subsoil assets (Japan and Hungary) and another seven countries had official estimates of other non-produced assets. So, a natural question to ask is whether wealth accounting has spread over these last 30 years?

Table 2.4 provides an overview of current country practices in national wealth accounting focusing on the real economy, with in the columns a breakdown by the types of non-financial assets covered.

It is important to stress that the scope of Table 2.4 is restricted in several ways. First of all, only countries who compile asset accounts – accounts that record opening and closing stocks and changes therein (such as depletion or discoveries or growth) during the accounting period – in monetary terms are included. For instance a country that compiles the value of extraction of timber but does not estimate the total stock of timber would be excluded. Secondly, only countries whose practices are considered to be part of, or related to, an official statistics program are included. Pilot studies conducted by academia without involvement of the statistics community are excluded. This does not make involvement of the statistical office a necessary condition; the Central Bank could play this role – as is often the case in Latin American countries. Even with these arguably strict criteria, not all country practices are clear-cut.

The table clearly has additional limitations. It has been constructed based on multiple sources, but no direct country survey was held. The method that was followed consisted essentially of three stages. First, existing surveys in different wealth accounting areas (subsoil, land etc.) were reviewed to draw up a first rough draft table. Secondly existing publications as well as the OECD statistics database were used to get a more precise picture. Third, country statistical offices' websites were visited in combination with follow-up interviews of country experts. However, due to non-response as well as language difficulties, for several entries the table primarily reflects the author's interpretation and judgment and is definitely open to further debate.

Table 2.4 Overview of country practices in wealth accounting for non-financial assets (2010)

Country	Minerals and energy	Timber	Fish	Land	Human Capital	Other assets[1]	Balance sheet produced assets	Natural capital included in balance sheet
Australia	R	R		R	P		yes	yes
Austria	P	P					yes	no
Belgium							yes	n.a.
Botswana	S						no	no
Brazil		I					no	no
Canada	R	R		R	P		yes	yes
Chile							yes	n.a.
Czech Republic	R			R			yes	yes
Denmark	R	P		P			yes	no
Estonia		P					no	no
Finland		R			P		no	no
France	R	P		R			yes	yes
Germany		R		P			yes	no
Guatemala		I					no	no
Hungary							yes	n.a.
Iceland			P				yes	no
India	P	P					no	no
Indonesia	R	P					no	no
Israel							yes	n.a.
Italy							yes	n.a.
Japan	R	R	R	R			yes	yes
Korea, Rep.	R	R		R			yes	yes
Mexico	R	R	I			yes	no	no
Namibia	S		S				no	no
Netherlands	R			P	I	yes	yes	no
New Zealand		P	R		P	yes	no	no
Norway	R	R	R		P	yes	other[2]	no
Philippines	S	S	S				no	no
Portugal							no	no
Slovak Republic				R			no	no
South Africa	S						no	no
Sweden		S	S				no	no
United Kingdom	R		P				yes	no
United States	S			S	P		yes	no
Total	18	19	7	11	7	3	19	6
of which regular	12	7	2	7	0	0		

Multiple sources: OECD Stats database (accessed 23 Feb. 2010); Searchable Archive of Publications on Environmental-Economic Accounting (http://unstats.un.org/unsd/envaccounting/ceea/archive); Results from the Global Assessment of Environment Statistics, Environmental-Economic Accounting and related statistics (UNSD, 2007); websites of NSOs; personal interviews.

Legend: R = accounts published on a regular basis (e.g. annually); I = accounts recently initiated but without results yet; P = accounts compiled as a pilot project that has not yet been taken into regular production; S = accounts compiled regularly in the past but currently suspended; Blank cell = no accounts initiated; n.a. = not applicable.

[1] For example, other forest asset values, water, and hydroelectric power.

[2] Norway publishes annual comprehensive wealth accounts but not as part of official balance sheets.

There are currently more than 30 countries that have compiled wealth estimates of which 16 are regular compilers of at least one type of natural capital stocks. The great majority of countries use the SEEA as reference handbook. In terms of types covered, timber and subsoil accounts have been compiled most often followed by land accounts. Produced assets followed by subsoil assets are compiled most regularly by countries. We will now discuss compilation practices in more detail by type of resource.

Mineral and energy accounts

Among the natural capital accounts, stock accounts for mineral and energy resources are compiled most regularly. Table 2.5 provides some characteristics for regular compilers of mineral and energy asset accounts. It has been constructed based on multiple sources and to some extent reflects the author's interpretation as no direct country survey was held.

The net present value method (see Chapter 3) is widespread although some countries – often developing countries – use the easier to implement net price method or the El Serafy method. Japan uses the Hoskold or sinking-fund method while the Czech Republic estimates stock values as the residual value of the stock of tangible non-produced assets minus the stock of land which are both available from statistical surveys (OECD, 2008).

Country practices differ regarding the assumptions used in the application of the NPV method: the chosen discount rates are often around 4% but rates of return vary between 4% and 8%. Canada calculates several variants of the NPV method resulting in upper and lower boundary values. The available time series vary across countries and some countries do not compile physical stock accounts. Australia and Norway, in 2010, appeared to be the only two countries that also publish stock values in constant prices.[5)]

[5)] Statistics Netherlands at present also publishes energy asset accounts in constant prices.

Table 2.5 Country practices in mineral and energy asset accounting (2010)

Country	Subsoil	Energy	Minerals	Valuation Method	Constant prices	Time series avaliable
Australia	yes	yes	yes	NPV	yes	> 15
Canada	yes	yes	yes	NPV (variants)	no	> 40
Czech Republic	yes	.	.	Other	no	> 5
Denmark	yes	yes	no	NPV	no	> 15
France	yes	.	.	NPV	no	> 20
Indonesia	yes	yes	.	Net price	no	> 10
Japan	yes	.	.	Hoskold	no	> 40
Korea, Rep.	yes	yes	yes	NPV	no	> 5
Mexico	yes	yes	.	Net price and El Serafy	no	> 10
Netherlands	yes	yes	yes	NPV	no	> 15
Norway	yes	yes	no	NPV	yes	> 25
United Kingdom	yes	yes	no	NPV	no	> 30

Multiple sources: a.o. Global Assessment of Energy Accounts (UNSD, 2009); personal communication.
Legend: . = unknown; > x implies that at least a time series of x years was found.

One of the main findings of the Global Assessment on Energy Accounting (UNSD, 2009) was that in all responding countries the total stock of reserves that is valued is broader than mere proven reserves as recommended in 2008 SNA. They may include for instance also probable reserves, which renders making comparisons across countries of reserves difficult. Another finding was that the main difficulty in applying the NPV method is fluctuating resource rents. Some countries therefore use a weighted moving average to smooth the effect of price changes, while others use specific price forecasts.

Timber accounts

Although around 20 countries have compiled timber stock accounts, only seven of them have turned into regular compilers. Possible explanations are that forests in addition to timber production often provide a broad range of services that due to their non-market nature are difficult to value. Many timber rich countries have chosen to compile economic accounts for forestry which provide information about the importance of the forestry sector for the economy, rather than pursuing stock accounts per se. The situation may change due to the increasing interest in ecosystem accounting (as will be discussed in Chapter 4).

Land accounts

Recently there has been an increase of interest in estimating stock values of land (Australian Bureau of Statistics, 2010; Statistics Netherlands, 2010c). At least eleven countries currently compile estimates for land but not all of them cover all types of land, and only six of them currently include these estimates in the national balance sheet. Several methods are used ranging from business surveys, and registers to household budget surveys (Kim, 2008).

Fish accounts

New Zealand and Japan appear to be the only two countries that regularly compile stock values for various species of fish. The Japanese estimate is based upon the capitalization method. New Zealand's valuation method uses quota sales for which a large competitive market exists. Several countries have experimented with fish accounts but when no quota valuation is possible the NPV method often proves difficult to apply in practice. Several pilot studies have resulted in negative resource rents that might be caused by strong vertical integration of the fisheries industry (Harkness and Aki, 2008).

Human capital

There is an increasing interest in the compilation of human capital accounts with at least seven countries that in 2010 had conducted pilot studies or just initiated work in this field. Most countries estimate human capital as the present value of future labor income using as the Jorgenson and Fraumeni lifetime labor income method (Jorgenson and Fraumeni, 1989) although differences exist regarding its precise application and scope (e.g. ages covered; treatment of non-market activities). In 2010 only three countries had compiled complete stock accounts and none compiled these accounts regularly. The OECD has established a consortium of countries with an interest in developing human capital accounts (OECD, 2011c), and estimates have been made for 15 countries. Statistics Netherlands (2012a) has recently published experimental human capital accounts (see Chapter 5).

Other stock accounts

New Zealand has experimented in the past with a stock account for water. The Netherlands has estimated within a pilot project stock values for renewable energy (wind and solar). Mexico has calculated the depletion of groundwater resources based upon a calculation of the shadow price of groundwater according to the residual value method in combination with an annual water balance.

Balance sheets

Although many countries have estimates for financial assets and liabilities, around 20 publish balance sheets for non-financial assets (covering at least produced assets).[6)] Only six countries include estimates for non-produced assets in their national accounts balance sheets. Norway compiles stocks of produced capital but these are not included in the balance sheets of the national accounts. Norway has a long research tradition on wealth accounting and it publishes an indicator of national wealth which is disseminated as part of their annual report on sustainable development indicators.

Country experiences with 'green GDP'

'Green GDP' – the attempt to modify or replace conventionally measured GDP or NDP by correcting for depletion and/or degradation of natural capital – has attracted controversy since its inception. As 'green GDP' has the connotation to many as aspiring to replace conventional GDP, it is often better to use the more neutral term environmentally adjusted aggregates to stress that in most countries the objective is not to replace GDP/NDP itself but to present alternative aggregates as part of their environmental satellite accounts. The plural 'aggregates' also better reflects that it is not necessarily GDP that is adjusted but more often NDP or the whole sequence of accounts including income and/or savings aggregates.

Table 2.6 provides an overview of country practices in the calculation of environmentally adjusted aggregates. The table needs to be interpreted with caution. Firstly, its scope is restricted to the realm of official statistics in order to assess practices by countries thereby excluding many academic publications or assessments performed by international organizations. Secondly, the table was compiled on the basis of a literature survey, hence no direct survey was held, therefore due to language issues, some countries may have been missed. The same disclaimer applies that for several entries the table primarily reflects the author's interpretation and judgment which is definitely open to further debate.

6) There are possibly additional non-OECD countries that compile balance sheets for produced assets whose practices were outside the scope of this assessment.

Table 2.6 Selected country experiences with environmentally adjusted aggregates (2012)

Country	Status	Depletion	Degradation	Time series (at least)	Adjustments of GDP/NDP/NNI
Australia	R[1)]	yes	yes	02-03/07-08	< 0.5%
China	S	no	yes	2004	3%
India	P	yes	yes	2003 (and others)	various studies
Indonesia	P	yes	yes	2000-2010	around 20%
Japan	S	yes	yes	1992-2000	about 1 %
Korea, Rep.	P	yes	yes	1985-1992	about 3%
Mexico	R	yes	yes	1985-2011	around 10%
Netherlands	R[1)]	yes	no	1990-2011	about 1-2%
Philippines	S	yes	yes	1988-1994	various studies (range from 13% towards 0%)
Taiwan	P	yes	yes	1996-1998	around 3%
Sweden	P	yes	yes	1991, 1993, 1997, 1999	various studies (range from 2% towards positive adjustments)
United States	S	yes	yes	1947–1991	> 8% (in 1987)

Multiple sources: a.o. Searchable Archive of Publications on Environmental-Economic Accounting (http://unstats.un.org/unsd/envaccounting/ceea/archive); Alfsen et al., 2006.
Legend: R = regular; P = pilot; S = suspended.
1) Experimental.

Among the 12 countries identified here that have experimented with adjusted aggregates, currently only three countries regularly compile environmentally adjusted aggregates. Even for these countries the adjustments made are certainly not exhaustive in terms of the scope of assets covered. The Netherlands has published a depletion adjusted income measure which takes the consumption of its energy reserves into account (Statistics Netherlands, 2010, 2011, 2012). Australia publishes a depletion-adjusted GDP that corrects for the depletion of subsoil assets and degradation of land, although it explicitly warns that the estimates are experimental.[7)] Mexico publishes an indicator called Net Ecological Domestic Product (or "PINE", its Spanish acronym) for several years now, which corrects for depletion and degradation. The Mexican National Institute of Statistics, Geography and Informatics (INEGI, by its Spanish acronym) is in fact mandated by the Mexican General Law of Ecological Equilibrium and Environmental Protection (LGEEPA, by its Spanish acronym) to compile this indicator on an annual basis.

Several countries have conducted pilot studies. Indonesia has a long experience with 'green GDP' starting with estimates by Repetto for the years 1971–1984 (Repetto et al., 1989). It has reported difficulties with valuation and data avail-

7) www.abs.gov.au/AUSSTATS/abs@.nsf/Lookup/4613.0Chapter25Jan+2010 (accessed March 2010); see also ABS, 2001.

ability (BPS, 2009). In a recent project, 'green GDP' has been estimated for the period 2000–2010, also with a regional disaggregation.[8] Taiwan has estimated environmentally adjusted aggregates for several years (DGBAS, 2002). For the Philippines several estimates of environmentally adjusted aggregates have been compiled in the past (Bartelmus, 1999), also according to rival accounting conventions (Peskin and Delos Angeles, 2001). In recent years, India has commissioned a series of pilot studies focusing on different regions and sectors covering a diverse range of adjustments as well as different valuation methodologies (cost and damage based) (UNSD, 2008). *"India says aims for green GDP alternative by 2015"* was a headline in October 2009 by Reuters India.[9] As a further step, in 2011, the Expert Group on greening India's National Accounts was established tasked with developing a framework and a subsequent implementation strategy. However, in its executive summary (PIB, 2013), the Expert Group has come out strongly against the concept of 'green GDP' which it calls a misnomer, and indicated it favors wealth accounting.

Several countries have suspended their environmentally adjusted aggregates measurements due to different reasons. In 2006 China for instance published 'green GDP' figures for 2004, which estimated damages at about 3% of GDP (SEPA and NBS, 2006). These estimates were lower than expected, and led to fierce debate. As a result, China seems to have reoriented its environmental accounting activities towards compiling specific types of accounts. Japan, after many year of following the SEEA 1993 methodology based upon the maintenance costs approach to value environmental pressures, decided due to measurement difficulties to shift its program towards accounting for pressures in physical terms only (ESRI, 2006). Furthermore, as is well-known, the United States' environmental accounting program has been suspended by Congress, shortly after its first publication in 1994. In Sweden, the National Institute of Economic Research has published several pilot studies that adjust aggregates (Skånberg, 2001; Gren, 2003). Gren uses a model following an ecosystem services approach that resulted in a positive adjustment of NDP due to transcending the production boundary of the national accounts. Ahlroth (2000) estimates adjustments based on an analysis of the effects of sulphur and nitrogen deposition. However, due to difficulties with valuation, these pilot projects never resulted in a regular publication of adjusted aggregates for Sweden.

In terms of scope of assets covered as well as valuation methods followed, there is wide diversity across countries. Some countries only subtract the costs of deple-

[8] http://www.unep.org/greeneconomy/Portals/88/documents/INDICATORS%20PPT/d3s3%20Gustami%20INDONESIAN%20EXPERIENCE%20IN%20DEVELOPING%20SUSTAINABLE%20DEVELOPMENT%20INDICATORs.pdf accessed May 2013.
[9] http://in.reuters.com/article/topNews/idINIndia-43127920091013 (accessed March 2009).

tion, others correct also for degradation costs. Some – often earlier pilot studies – closely follow the SEEA 1993 guidelines[10], whereas others follow the more recent SEEA 2003 or SEEA CF. Others departed from SEEA guidelines, for instance the United States, in its treatment of discoveries of subsoil assets, or proposed alternative frameworks (Peskin and Delos Angeles, 2001).

Valuation methods for degradation obviously depend on the type of damage that is estimated. Both cost based estimates (such as maintenance costs, restoration costs etc.) as well as damage based estimates are used. Due to various valuation difficulties as well as the absence of clear international standards, many countries have used various methodologies and assumptions to assess sensitivities of their results to assumptions and have given ranges of estimates rather than precise numbers. As shown in Table 2.6, the adjustments that have been published range from double digit figures towards positive adjustments. However, due to differences in assets covered as well as the type of aggregate (GDP, NDP etc.) that is being adjusted, it is not very meaningful to compare these estimates across countries. Most countries have developed time series that allow for comparisons over time. Not included in the table are several pilot studies that were conducted by research agencies or by international organizations, for instance in the early 1990s in amongst others Costa Rica, Ecuador, and Papua New Guinea. Another well-known research project is the Green Accounting for Indian States Project[11] which uses a 'top-down' macroeconomic approach to model various adjustments to GDP with a regional breakdown for all India's states for 2003.

In the academic community several research projects have been carried out which attempt to estimate externalities for a wide range of countries. GARP[12] (and its successor GREENSENSE) has primarily focused on the impacts of air pollution based upon impact pathway assessments with the ECOSENSE model. The impacts on human health, crops and building materials were estimated and attributed to four countries expressed as a percentage of GDP and subsequently disaggregated to economic sectors (Markandya, 2000). Whereas GARP uses a damage based approach, the GREENSTAMP[13] project quantifies 'economic opportunity costs associated with meeting specified environmental performance standards' which would results in a "greened GDP" 'an estimate of the level of output … that a national economy would be able to achieve while simultaneously respecting specified

[10] Compared to its successor SEEA 2003, SEEA 1993 had a stronger focus on the derivation of environmentally adjusted aggregates (it distinguished for instance between EDP1 and EDP2) and clearly advocated the maintenance costs approach to evaluate costs of degradation.
[11] http://www.gistindia.org/
[12] Green Accounting Resource Project I and II.
[13] GREEned National STAtistical and Modelling Procedures.

environmental … requirements.' (Brouwer et al., 1999). More recent projects such as EXIOPOL[14] and its successor CREEA[15] as well as EORA[16] integrate research on externalities within an environmentally extended Multi-Regional Input-Output framework. Although these recent initiatives do estimate costs of depletion and/or degradation, which are required ingredients for compiling an environmentally adjusted aggregate, the focus is not on estimating 'green GDP' as such. Likewise, the World Bank estimates costs of depletion and degradation but uses these for estimating its adjusted net savings indicator.

Some countries have come out explicitly against 'green GDP' measures. Well-documented is the German experience where a scientific Advisory Committee was established that over the course of 12 years discussed the concept of 'green GDP' extensively and concluded that *"views on a green GDP have changed over time and how – because it was not feasible – it became increasingly clear that such a variable actually has more the character of a model"* (Advisory Committee, 2002). A similar conclusion was drawn in Norway which did consider compiling 'green GDP' in the mid 1990s but in the end considered it 'impractical' (Alfsen et al., 2006).

2.5 Conclusions

From the assessment of country experiences in Section 2.4, we conclude that environmental accounting in general is clearly a growing area of statistics. Environmental accounting programs have been established in all regions of the world, both developed and developing. It is fair to say, that there is not a single country that compiles all types of environmental accounts, although several countries such as Norway, Sweden, the Netherlands, Canada have come close in recent years. The development of environmental accounting is often guided by country-specific circumstances. For resource abundant countries, natural resource accounts may have a higher priority, while in countries with high pollution rates, water or air emission accounts may be more policy relevant.

[14] EXIOPOL: A new environmental accounting framework using externality data and input-output tools for policy analysis; www.feem-project.net/exiopol
[15] CREEA compiling and refining environmental and economic accounting http://creea.eu/
[16] www.wordmrio.com

Compared to our benchmark year of 1979, we have seen that wealth accounting has become increasingly widespread in the statistical community. The strongest advances have been made in the areas of mineral and energy accounting while recent years have seen an increasing interest in assessing stock values of land and human capital. The valuation of renewable assets (like fish or water) lags behind due to measurement difficulties. At the same time, international agencies such as the World Bank (2006; 2011) and more recently UNU-IHDP and UNEP (2012) in its inclusive wealth report, have estimated wealth for individual countries (World Bank, 2011 covers 120 countries; UNU-IHDP and UNEP, 2012 covers 20 countries). This raises the question how these estimates compare with official statistics. This topic will be further explored in Chapter 5 were we will review and compare the World Bank's wealth estimates for the Netherlands with official data.

The experience of countries with 'green GDP' shows a more mixed picture. The experiences range from countries such as Mexico which are mandated by law to publish a 'green GDP' indicator, towards countries that have come out strongly against 'green GDP' measures. It is clear that the concept 'green GDP' still inspires, as may be evidenced by the fact that in the last couple of years several countries announced 'green GDP' initiatives. For example, ahead of the Rio+20 Conference Denmark announced its ambition to introduce a 'green GDP'[17] and also Vietnam is developing a 'green GDP'.[18] However, the percentage of countries to date that has mainstreamed the compilation of adjusted aggregates remains low. We will come back to this issue in Chapter 7.

With the background and context on environmental accounting that was provided here, we are ready to investigate theory and practice in various domains in greater detail in the next chapters.

[17] http://www.dr.dk/Nyheder/Andre_sprog/English/2012/06/19/130352.htm (accessed July 2012).
[18] http://vietnamnews.vn/Industries/222756/nation-targets-green-growth.html (accessed July 2013).

3. Depletion: bridging the gap between theory and practice

3.1 Introduction[1)]

In this chapter we investigate empirical and theoretical approaches towards estimating costs of depletion that have been recently proposed within the statistical and research communities. In the System of National Accounts (SNA) (UN et al., 2009) the income derived from the extraction of non-renewable resources such as oil or minerals is recorded in full in the current accounts, without charging for the cost of depletion. As explained in Chapter 1, when the depletion of resources hampers future production possibilities, indicators such as net national income when unadjusted for the cost of depletion yield incorrect signals to policy makers about future welfare. It is therefore of crucial importance that the measurement of depletion is satisfactorily resolved.

These criticisms are well known (Harrison and Hill, 1994; Vanoli, 1995) and increasingly recognized, for instance within the GDP and Beyond Roadmap (European Commission, 2009) and by the Stiglitz-Sen-Fitoussi report (Stiglitz et al., 2009), which have both called for indicators to complement Gross Domestic Product (GDP) taking amongst others depletion into account. Nevertheless, disagreement remains about how depletion costs should be defined, valued and recorded.

In the statistical community, accounting for the depletion of natural resources is one of the prime motivations for the System of Environmental-Economic Accounting (SEEA), a satellite system to the SNA. The SEEA Central Framework, which was adopted in February 2012 as an international statistical standard (UN et al., 2012), contains for the first time a unique recommendation on the measurement and accounting treatment of depletion and proposes several depletion adjusted aggregates such as income and savings. The proposed depletion measure differs from depletion as change in total wealth that was proposed in the SEEA 2003 (and which also underlies the El Serafy method (El Serafy, 1989)). For statisticians, underneath the issue how to properly cost depletion, lies the accounting issue how depletion of non-renewable resources should be understood and subsequently recorded in an accounting framework. This issue seems to have received little attention by the theoretical community.

1) This chapter is based upon Edens (2013a). The author would like to thank Christian Bogmans, Kirk Hamilton, Taoyuan Wei, and Cees Withagen for commenting on an earlier draft of this article. Also the helpful comments by two anonymous referees are much appreciated. The author would also like to acknowledge Reyer Gerlagh who provided various suggestions for improvement and clarification of the draft chapter. The author is also indebted to discussions with Ole Gravgård Pedersen. The introduction also draws upon Edens and Van Rossum (2012).

In the theoretical community, accounting for non-renewable resources has a long tradition (the seminal articles are Hotelling, 1931; Dasgupta and Heal, 1974; Weitzman, 1976). Recently, several new depletion measures have been proposed. Following the work by Sefton and Weale (2006), Asheim and Wei (2009) have proposed a theory called sectoral income in which depletion is defined as net investment. Hamilton and Ruta (2009) have proposed to measure depletion as net savings, based on the framework of comprehensive wealth associated with the work of Arrow et al. (2003a). Their approach has been used in recent World Bank estimates for wealth (World Bank, 2011). Cairns (2009) is another recent contribution, that argues for basing depletion estimates on microeconomic foundations instead of on conventional macro models.

The purpose of this chapter is to bridge the gap between theory and practice. We will do this by clarifying definitions and assumptions for the main theoretical and empirical approaches, and subsequently evaluating all proposed depletion measures using a data set of Dutch natural gas reserves. As is well known, the Netherlands has significant natural gas reserves that were discovered in the late 1960s. The sale of natural gas is an important annual source of income for the Dutch government. In 2011, revenues amounted to 12.4 billion €. Over the period 2001–2011, government revenues from natural gas sales have averaged 3.7% of total government receipts. The percentage ranged from 2.6% in 2002 to 5.4% in 2008. The importance of natural gas revenues also becomes evident, if the effect on the public deficit is taken into account. Including and excluding natural gas revenues, the government deficit was respectively 4.5% and 6.5% of the gross domestic product (GDP) in 2011 (Edens and Van Rossum, 2012). Adjustments for depletion expressed as a percentage of net national income are therefore highly policy relevant. Finally, we will address the question, which measure is appropriate when adjusting macro-economic aggregates such as Net Domestic Product (NDP) or Net National Income (NNI).

The outline of this chapter is as follows. In Section 3.2 we will provide a theoretical foundation for discussing depletion by discussing a firm engaged in extracting a non-renewable resource. We conclude that in practice estimating depletion is difficult because due to empirical realities such as non-optimality and uncertainty, we are forced to move beyond the simple theoretical model. Section 3.3 provides an overview of the discussion of depletion as it has taken place in the statistical community during the past two decades. The dominant view that has emerged considers the depletion of natural resources akin to the depreciation of fixed assets. In Section 3.4 we will discuss the four depletion measures introduced above and show how they relate to the basic model. In Section 3.5 we evaluate these four depletion measures on Dutch data on natural gas extraction. We find that correcting for the cost of depletion would lead to significant adjustments of both level of net national income (ranging between 1.6 and 2.4% for 2008) and growth rates of Dutch net national income in current prices (ranging from –1.1 towards +1.4% adjustment), with a strong dependency on the chosen measure. In Section 3.6 we will see that some criticisms of empirical accounting practices regarding depletion are misguided. We will argue that the choice for a depletion measure depends primarily on the context of use: measurement of social welfare, sustainable income, or consistency with the national accounts' income concept. Section 3.7 concludes.

3.2 Depletion: the classic theoretical case

In order to provide a theoretical foundation for estimating depletion costs, it is useful to start with the classic textbook example of a firm engaged in extracting a non-renewable resource. Following the notation of Cairns (2009, p.120–121), in a basic micro model (without exploration; price and interest rate exogenous) the objective of the firm is to maximize profits:

$$\max \int_t^T \big(p(s)q(s) - C(q(s))\big) e^{-r(s-t)} ds \tag{1}$$

subject to productive constraints: $q(t) = -\dot{S}(t) \geq 0$ and $S(t) = S$

with:

$p(t)$	price at time t;
$q(t)$	extraction at time t;
$C(q(t))$	the cost of extracting a quantity $q(t)$;
$S(t)$	the stock at time t;
T	time of exhaustion;
r	the interest rate.

The current value Lagrangean for this optimization problem is:

$$L = pq - C(q) - \lambda q$$

The first order condition for an interior solution is:

$$\partial L / \partial q = p - C'(q) - \lambda = 0$$

or

$$\lambda = p - C'(q) \tag{2}$$

Therefore, the shadow value of the resource (underground) λ consists of the market price of the product sold (above ground) minus the marginal cost of extraction. Furthermore we have that

$$-\partial L / \partial S = \dot{\lambda} - r\lambda = 0 \tag{3}$$

This results in the Hotelling rule (or r-percent rule) $\dot{\lambda} / \lambda = r$ which states that marginal profit (the shadow value of the resource) increases with the interest rate. The solution of the optimization problem results in an optimal path of future extractions $\left\{q^*(t)\right\}$.[2] The value of the mine $V(t)$ equals the discounted sum of future profits earned corresponding with this optimal path:

$$V(t) = \int_t^T \left(p(s)q^*(s) - C(q^*(s)) \right) e^{-r(s-t)} ds \tag{4}$$

By differentiating the right hand side of Eq. 4 we obtain that for a given extraction and price path depletion – the change in value of the resource – can also be written as the return to the asset minus the current profit (sometimes called net cash flow or rental):

$$\frac{dV(t)}{dt} = rV(t) - (p(t)q^*(t) - C(q^*(t))) \tag{5}$$

With prices exogenous, the value of the mine will in general be a function of both the remaining stock as well as time, again following Cairns (2009, p.123), we write $V(t) = Z[(S(t), t]$. Therefore we have that

$$\frac{dV(t)}{dt} = \frac{\partial Z}{\partial S}\frac{\partial S}{\partial t} + \frac{\partial Z}{\partial t} \tag{6}$$

Now, in case of a stationary economy, for instance when we make the additional assumption that profit depends on the level of extraction only

[2] The assumption of optimality which implies the validity of the Hotelling rule, places restrictions on the combination of prices and extraction costs: for instance a constant price level in combination with constant marginal extraction costs does not give an interior solution.

$\pi(q(t)) = p(q(t))q(t) - C(q(t))$ the problem becomes autonomous (see also Hartwick and Hageman, 1993).[3] From optimal control theory we know that the shadow price of the resource is equal to the increase in the optimal profit that would result from an increase in the stock i.e. $\lambda \equiv \frac{\partial Z}{\partial S}$. We therefore obtain that depletion for a stationary economy can be written as the current period's extraction multiplied by the shadow price:

$$\frac{dV(t)}{dt} = \frac{\partial Z}{\partial S}\frac{\partial S}{\partial t} = -\lambda q^*(t) \tag{7}$$

Eq. 7 expresses that for an optimal stationary economy depletion equals the current period's extraction multiplied by the profit on the last ton mined (the latter is called total Hotelling rent by Hartwick and Hageman (1993)). According to Hartwick and Hageman (1993) the total profits (in a certain accounting period) can therefore be understood to consist of two types of rent: total Hotelling rent and Ricardian rent (rV).

Table 3.1 Overview of interpretations of expressions

Expression	Interpretation	Other descriptions encountered in the literature
$\frac{dV(t)}{dt}$	Depletion	Current rent
$rV(t)$	Return to asset	Ricardian rent; income element; sustainable income
$\lambda q^*(t)$	Extraction times shadow price	Total Hotelling rent; extraction times profit on the marginal ton mined
$p(t)q(t) - C(q(t))$	Resource rent	Profit

Now it is important to clarify that in the SNA the word 'rent' (economic rent is used interchangeable with resource rent) is used in a specific meaning which may be different from how it is usually understood in environmental economics, as we described above.[4] To be precise, for the simple model introduced above (i.e. in the absence of labor, taxes, depreciation of fixed assets etc.) rent according to the SNA would be equal to profits or earnings i.e. $p(t)q(t) - C(q(t))$. We will follow in this paper the SNA definition of rent and in order to distinguish it from other notions of rent, refer to it as resource rent. In Section 3.4 we will discuss more precisely how

[3] This would correspond to a situation in which the firm would be a price setter.
[4] According to the 2008 SNA (UN et al., 2009) para. 20.47: "Suppose that a mining company knows the size of the deposit being mined, the average rate of extraction and the costs of extraction of one unit. After allowing for all intermediate costs, labour and the cost of fixed assets used, what is left must represent the economic rent of the natural resource. By applying this to the expected future extractions, a stream of future income can be estimated and from this … a figure for the value of the stock of the resource at any point in time."

it is defined in the SNA and how it is statistically observable. Finally, it is important to realize that Eq. 5 is valid for any given fixed extraction and price path, and does not require optimality.[5] This property is exploited in Section 3.4 when we move beyond the basic model.

What this basic theoretical model illustrates is that there are essentially two approaches towards estimating the cost of depletion that yield identical results: either we calculate depletion via the change in value of the resource (Eq. 5) or we estimate the shadow price of the resource multiplied by current extraction (Eq. 7). However, the equivalence is only true given the strict assumptions such as optimal use, stationarity, and fixed stock.

In practice, estimating depletion costs is difficult due to various issues:

- Market prices for operational mines/fields are usually not available as changes in ownership seldom occur due to the fact that investments are sunk or the existence of legal constraints. Direct estimates of $V(t)$ which would allow to evaluate Eq. 5 are therefore often unavailable.
- Data on marginal extraction costs are generally unavailable, which makes the total Hotelling rent unobservable. Cairns (2009, p.135) argues that in reality there are several constraints that are imposed on extraction (in addition to the productive constraints on Eq. 1) such as physical properties of the resource (e.g. sufficient pressure) or the existence of binding regulations. Each constraint will have a positive shadow price, which confounds the observable net price (or rent). In short, Eq. 7 is also often empirically unobservable.
- The assumption of optimality which underlies the basic theoretical model is questionable. In practice there will be various types of distortions (e.g. externalities). Moreover, there will be multiple firms and mines which may lead to problems of aggregation. The fixed stock assumption is invalid as there are discoveries and reappraisals. Moreover the concept of reserves in practice is not a physical concept but an economic concept, so the estimate of reserves itself is dependent on price expectations (as well as technology).
- In order to derive Eq. 7 we assumed a stationary economy, but in reality interest rates and prices will change through time which renders the problem non-autonomous and we have the additional term $\partial Z / \partial t$ as in Eq. 6.[6]
- Finally, the model discussed here is deterministic, but reality is uncertain and there may be all sorts of unexpected events (e.g. discoveries). It is not

[5] Eq. 5 is what Hartwick and Hageman (1993, p.214) have called the "fundamental equation of asset equilibrium"; Cairns (2009) refers to it as the "fundamental equation of economic accounting".

[6] The term $\partial Z / \partial t$ has been given various names in the literature: here we consider it an expression of the effect of time passing which is a form of a capital gain (see Usher, 1993) when the latter term is understood as a change in the value of an asset during the accounting period.

straightforward how best to model such unexpected changes, and a stochastic model may be required. Perman et al. (2003, p.526) provide an example in which the existence of a probability of a disaster (the reverse would be a probability of a discovery) would lead to an increase (decrease) in the discount factor.[7)]

Given these difficulties, it may come as no surprise that numerous methods have been proposed in the literature regarding estimating depletion costs at the national level that move beyond the theoretical model discussed above. We will discuss four methods in greater detail in Section 3.4: two methods proposed in the statistical community and two proposed by the theoretical community. For each method we will indicate how it relates to the basic theoretical framework that was presented here.

However, first we will turn towards the discussion of depletion by the statistical community. Apart from the issue of how to cost depletion, there is an arguably even more fundamental issue, how the depletion of non-renewable resources should be understood in the context of the SNA. This issue has not received much attention by the theoretical community, but has major ramifications for the recording of depletion in the accounts, and a fortiori for macroeconomic indicators.

3.3 Overview of recording of depletion as discussed in the statistical community

Discussion of depletion in the statistical community started already in the late 1980s when the first SEEA handbook (UNSD, 1993) was being developed. The objective at that time was to derive a 'green GDP' (in fact, a misnomer, as it should be a green NDP) which would 'correct' GDP for the costs of depletion and degradation, punishing countries with large negative environmental externalities. Although the SEEA 1993 handbook included an eco-domestic product, its successor the SEEA 2003 (UN et al., 2003) refrained from proposing a single indicator, and in fact allowed for multiple options in measuring depletion. With the development of SEEA Central Framework (UN et al., 2012), a unique recommendation for depletion

7) Gaudet (2007) also contains a discussion of the Hotelling rule in case of the presence of uncertainty.

has been proposed, that was recently adopted as an international statistical standard by the United Nations Statistical Commission which places it on par with the SNA.

One of the reasons why it has taken the statistical community such a long time to come up with a unique recommendation is that it is not clear how the depletion of non-renewable resources should be understood in an accounting framework. Should it be treated similar to the consumption of fixed capital, as the sale of inventories, or should resources be seen as becoming produced through the act of mineral exploration? While in theoretical models it is sufficient to specify that a non-renewable resource is considered an asset (or a form of capital), in accounting it becomes important what kind of asset we are discussing: produced or non-produced; stocks or inventories? Moreover, we are restricted by all sorts of conventions that apply depending on these characteristics.

In addition, we will try to characterize the main proposals by linking them to the theoretical framework developed in the previous section. The recording of depletion is related to the recording of income in the national accounts as well as the treatment of discoveries. From a theoretical perspective, the recording of income can be seen as an intertemporal budget allocation problem; in case of a non-renewable resource, how should its value (Eq.4) be allocated across multiple accounting periods?

How to understand and record depletion?

In the SNA (UN et al., 2009) the income derived from the extraction of non-renewable resources such as oil or minerals is recorded in full, and no costs of depletion are accounted for in the currents accounts. The 2008 SNA includes the value of the resource (see Eq. 4) in national income through the actual resource rents at the time they are generated. In case there are unexpected discoveries, the additional value would not be included in income of the period of discovery, but deferred to the time actual resource rents would be generated during extraction. The SNA does in fact record depletion in the balance sheets, but not in the production or generation of income account. This view can only be upheld if natural resources are infinitely abundant and hence do not change in economic value over time. However, this position is generally considered to be empirically untenable.

With this in mind, Table 3.2 provides an overview of the main recording proposals for depletion that have been made in the statistical community, in terms of key characteristics and their impacts on GDP and NDP.

Table 3.2 Categorization of approaches towards depletion

	2008 SNA	El Serafy	Vanoli	BEA/Repetto	SEEA Central Framework
Interpretation	Abundant	Sale of asset	Sale of asset	Produced	Fixed assets
GDP (includes)	R	R–D	–	R+A	R
NDP (includes)	R	R–D	–	R+A–D	R–D

Source: Vanoli (2005, p.339-341) with modifications and additions.
R = Resource rent; A = Discoveries; D = Depletion (expressed here as a positive value).

Vanoli (1995, p.127–129) argues that revenues from the extraction of subsoil reserves should not be considered as income from production. Seen as an inter-temporal budget allocation problem, Vanoli's position seems to be that the value of the resource has already been counted in the past, and should not be included in income again. He reasons that subsoil reserves are not created during exploration, they are rather pre-existing assets ('gifts of nature' according to Vanoli, 2005), that during the production process are merely relocated and traded. Depletion of these reserves can therefore not be treated as consumption of fixed capital as the reserves are not used as assets in a production process of other goods. He argues that the production process of oil and gas has more in common with industries such as wholesale. The correct accounting treatment therefore should be as 'withdrawals from inventories'. According to Vanoli, the value of these withdrawals should not be treated as intermediate consumption, but deducted from output in the production account, in a way similar to the recording of the production of trade margins in the national accounts. These withdrawals should accordingly be valued using the rent. As a result, the value of rent would be excluded from both GDP and NDP.

El Serafy's 'user cost approach' (El Serafy, 1989) is conceptually close to Vanoli, as both consider rent as the "proceeds of the sale of non-produced assets" (Vanoli, 2005, p.340). But in contrast to Vanoli, El Serafy partitions the rent into an income and a depletion element. The basic intuition is that part of the rent should be invested to allow a perpetual annuity. The size of the income element can be calculated, by imposing that its net present value over an infinite period has to be equal to the net present value of the original resource.[8)] An important difference between El Serafy and some of the other proposals is that El Serafy proposes to also deduct a cost of depletion from GDP (and not just from NDP).

8) In calculating depletion costs, El Serafy's formulas are a special application of a Net Present Value model, with a constant extraction rate and fixed interest rate.

The view advocated by Repetto and the United States' Bureau of Economic Analysis (BEA) (Nordhaus and Kokkelenberg, 1999) and recently also by Cairns (2009) is by contrast that the value of oil and gas reserves is in a sense created by the economic activity of mineral exploration. They should be therefore considered as produced assets, rather than as non-produced assets as is common in the SNA. As a result, the value of discoveries should be included in GDP. The main intuition of this position is that before depreciating an asset one should first properly recognize it as an investment. In terms of the intertemporal budget allocation problem, the value of the resource is included in income at the moment of discovery.

The 'mainstream' view that eventually was put forward by the statistical community and that also forms the basis of the SEEA Central Framework is that the depletion of non-renewable resources should be treated similar to the consumption of fixed capital (Comisari, 2008). The resource rent is included in full in GDP, but in the estimation of NDP a cost of depletion should be deducted which represents the depreciation of the resource (sometimes referred to as consumption of natural capital). The cost of depletion can be obtained by splitting the resource rent into a depletion and income element (Obst, 2010) as in Eq. 5. Interpreted as an intertemporal budget allocation problem, the main idea being that it is the return to the asset (resource rent corrected for depletion) that should be included in income (as in the El Serafy approach).

Discussion between approaches

The consumption of fixed capital approach to depletion was eventually favored due to a number of reasons that are well summarized in Obst (2010). The key argument is that there is a generic similarity between consumption of fixed capital (e.g. machines) and the depletion of non-renewable resources (subsoil deposits) as both of them are used up in the production process over a long period of time. This characteristic outweighs notable dissimilarities between natural resources such as that (1) natural resources are non-produced assets whereas fixed assets are by definition produced, and (2) while fixed assets necessarily depreciate with time – even when left unused – because of obsolescence, oil deposits do not automatically deplete in time and may even appreciate in value.

In addition, the alternative proposals that were put forward also face severe shortcomings. The main argument against the interpretation of the sale of an asset is that oil deposits, contrary to standard inventories, are not available for sale immediately, at least not the whole deposit. More importantly, as argued by Obst (2010, p.3), when the entire resource rent is deducted from income, we may run into a number of accounting problems that would make this approach difficult to implement in practice. Moreover, when the entire resource rent is removed from output and value added, extractive industries would essentially be reduced to whole sellers. According to Obst (2010, p.4) this would "distort the financial reality" in which these industries may generate high incomes and provide a source for government revenue. Such an accounting treatment would therefore "reduce the utility of the national accounts because the proposals are so far removed from generally accepted business and government accounting principles" (ibid.).

The proposal to conceive of the discovery of subsoil reserves as capital formation and hence as adding to GDP would imply that time series of GDP would become volatile and undermine the usefulness of GDP growth rates as key indicator. Also its link with movements in production becomes obscured (Nordhaus and Kokkelenberg, 1999).[9)] Indeed, the SEEA Central Framework (UN et al., 2012) proposes to treat only the mineral and exploration costs as an intellectual property asset rather than the full discovered resource (which is entered through the other changes in volume). The intellectual property asset is subsequently depreciated and treated as a user cost of capital.

Finally, as argued by Obst (2010, p.4), treatment of fixed assets "appears to send the appropriate message to policymakers ... that ... depletion of a non-renewable natural resource over time will have an increasing negative impact on NDP". Although the choice for interpreting depletion akin to consumption of fixed capital does settle the key recording issues, it does not specify a unique recommendation how depletion costs should be assessed, which we will discuss now.

3.4 Four alternative depletion measures

In this section we fill discuss four alternative depletion measures. With respect to Table 3.2 these four measures all result in a depletion estimate D which can be recorded as in the SEEA Central Framework proposal as a charge against NDP.

9) According to Vanoli (2005, p.339) the BEA reasoning may have been influenced by the fact that the United States' national accounts (the NIPA) differ from the standard SNA as they do not include an account for "other changes in the volume of asset accounts" through which discoveries usually enter.

In the absence of directly observable market prices for assets (in our case oil and gas reserves), the SNA stipulates that an attempt must be made to estimate what prices would prevail if the assets were to be traded on a specified date. For valuing environmental assets indirectly, the preferred method is to estimate values based upon the net present value (NPV) of future earnings.

What is statistically observable is the gross operating surplus (i.e. profit in ordinary parlance) of the companies involved in oil and gas production. From the gross operating surplus, by deducting the user cost of fixed capital (which consists of depreciation, taxes less subsidies, and the opportunity cost of investments) we obtain the resource rent as defined in the SNA (UN et al., 2009) and SEEA Central Framework (UN et al., 2012). When additional assumptions about future prices, extraction paths, and the discount rate are made, indirect estimates of $V(t)$ can be obtained. Stock estimates are necessary in order to calculate depletion costs for the four different methods that we will now introduce formally.

Depletion as change in total wealth

The value of a fixed asset can be estimated as the net present value of the expected stream of future earnings (Eq. 4). In national accounts we do not work in continuous time. In discrete time and for ease of exposition, Eq. 5 can be written as:

$$V_{t+1} = (1 + r_t)V_t - R_t$$

with

$V_t \equiv V(t)$ the stock value of the resource at time t;

R_t the resource rent at time t;[10)]

r_t the time specific discount rate.

Rearranging yields that the resource rent can be split into a return to the asset and a change in value term that occurred during the accounting period:

$$R_t = r_t V_t + (V_t - V_{t+1}) \equiv r_t V_t + D_t \tag{8}$$

where we have defined depletion as the change in value during the year i.e.

[10)] In fact R_t is a refinement of the term $p(t)q(t) - C(q(t))$ in the sense that the cost function takes all costs into account including costs of labor, of exploration and user costs of capital. To be precise, according to the SEEA Central Framework (UN et al., 2012, p.154-155), the costs of exploration are considered as an investment (gross fixed capital formation) whose output consists of an intellectual property product. The user costs of this produced asset are deducted in order to obtain the resource rent.

$$D_t \equiv V_t - V_{t+1} \tag{9}$$

Although Eq. 8 is the discrete analogue of Eq. 5, it is important to stress that in order to derive Eq. 8 we did not require assumptions about optimality or the validity of the Hotelling rule that underlies the basic theoretical model discussed in Section 3.2.[11)]

Depletion as 'using up' of the resource

An alternative depletion measure that was eventually adopted in SEEA Central Framework (UN et al., 2012) is to define depletion in a physical sense as the cost of 'using up' the resource. In this measure, depletion is explicitly grounded in changes that occur during the accounting period in physical stocks S_t due to extraction.

The idea is as follows. First, stock data in physical terms can be used to decompose the opening stock value as a price multiplied by a volume as:

$$V_t \equiv \frac{V_t}{S_t} S_t = p_t^{gr} S_t \tag{10}$$

where

p_t^{gr} the price of one unit of resources in the ground; and

S_t the stock of the resource (underground).

The change in value during the accounting period can be decomposed into two elements: a depletion component and a revaluation component, as follows.

$$V_{t+1} - V_t = 1/2\left\{(p_{t+1}^{gr} + p_t^{gr})(S_{t+1} - S_t) + (S_{t+1} + S_t)(p_{t+1}^{gr} - p_t^{gr})\right\} \tag{11}$$

This results in a physical definition of depletion as 'using up' of the resource:

$$D_t^{phys} = 1/2\left\{(p_{t+1}^{gr} + p_t^{gr})(S_{t+1} - S_t)\right\} \tag{12}$$

Eq. 12 defines depletion as the change in physical stocks that occurred during the accounting period due to extraction, multiplied by the average price of the resource in the ground. Obviously, in addition to extraction, changes in physical stocks can be also due to discoveries and reclassifications, which are valued with the same

11) When the Hotelling rule would be followed in valuing the stock, its value would be equal to the unit resource rent times total physical stock as future rents rise with the discount rate (this type of valuation is called Hotelling valuation or net price valuation in SEEA 2003 (UN et al., 2003, p.282).

average price in order to derive a complete stock account (as will be shown in Section 3.5).

When we compare this approach with our theoretical model of Section 3.2, we see that this average price $1/2(p_{t+1}^{gr} + p_t^{gr})$ can be seen as an approximation of λ (Eq. 7).

It is important to stress that the two approaches discussed so far use different prices. The price of one unit of the asset in the ground p_t^{gr} is calculated by dividing V_t by S_t. By contrast, the resource rent is usually written as the product of the unit resource rent p_t^{rr} and the quantity extracted i.e.:

$$R_t \equiv \frac{R_t}{q_t} q_t = p_t^{rr} q_t \tag{13}$$

with q_t the quantity extracted. It should be stressed that both p_t^{rr} and p_t^{gr} are different from the p_t^{mkt} the actual market price.

Depletion as net saving

The theory of comprehensive wealth is associated with the work of Weitzman (1976) and more recently with Arrow et al. (2003a). Comprehensive wealth can be defined as the sum of all capital stocks of an economy evaluated at their shadow prices (e.g. Dasgupta, 2009). The main attraction of this method stems from the fact that under certain assumptions changes in comprehensive wealth correspond to changes in social welfare. According to the theory of comprehensive wealth, the depletion cost of a non-renewable resource is obtained by multiplying the shadow price of the stock of resources – defined as the partial derivative of the value function with respect to an additional unit of the resource – with the current extraction rate:

$$D_t^{cw} = \frac{\partial W_t}{\partial S} q_t = p_t^{sh} q_t \tag{14}$$

with
W_t value function at time t.

Based on the comprehensive wealth approach, the World Bank (2011) has recently updated its method for estimating depletion.

Hamilton and Ruta (2009) illustrate the approach in a Dasgupta-Heal type economy with a finite stock of resources S that is extracted at a constant rate q until exhaustion at time T, with extraction costs $C(q)$. The social planner chooses an allocation mechanism which ensures that q remains constant as well as the unit resource rent p^{rr}. Utility is assumed to depend only on consumption G. We therefore have (following Hamilton and Ruta, 2009):

$$F(K,q) = G + \frac{dK}{dt} + C(q) \tag{15}$$

with

F the production function;

K the capital stock;

G consumption.

We have:

$$S(t) = (T-t)q$$

$$p^{rr} = F_q - C(q)/q$$

The social welfare function consists of the discounted future consumption and is equal to the total stock value, which consists of both produced and natural capital:

$$W_t \equiv K_t + V_t = \int_t^{\infty} G(s)e^{-r(s-t)}ds \tag{16}$$

We can easily show that:

$$p_t^{sh} = \frac{\partial W_t}{\partial S} = \frac{\partial V_t}{\partial S} = \frac{p^{rr}}{T-t}\int_t^T e^{-r(s-t)}ds = \frac{V_t}{S_t} \tag{17}$$

The shadow price p_t^{sh} can be calculated endogenously as the stock value divided by the physical stock, and depletion can be estimated based upon Eq. 14. As in the previous approach, p_t^{sh} can be seen as an estimate of λ (Eq. 7), however it is now grounded in a macroeconomic model. We note that for this type of model, p_t^{sh} coincides with p_t^{gr}. It is important to stress that the shadow price depends on the model that is assumed, for instance when the utility function would depend directly on the resource stock (e.g. amenity value), the shadow price would be no longer equal to the asset price.

Depletion as net investment

Asheim and Wei (2009) have developed an alternative green accounting theory called sectoral income. The theory is based upon a model in which the instantaneous change in welfare is represented by net investments. These investments are based on changes in the present value of future commodity flows to a specific sector. Although the theory is formulated in continuous time, Wei (2009) applies the sectoral income theory to the Norwegian petroleum sector which requires discretisation.

We can derive the sectoral income theory in a discrete setting as follows. First, rearranging Eq. 7 and assuming a constant interest rate r (for ease of exposition) we obtain:

$$rV_t = R_t^{si} + V_{t+1} - V_t = R_t^{si} + \sum_{i=t}^{\infty} \frac{1}{(1+r)^{i-t+1}} [R_{i+1}^{si} - R_i^{si}] \tag{18}$$

with
R_t^{si} current cash flow as defined in sectoral income theory.

In sectoral income theory, the sectoral income r_tV_t (left hand side of Eq. 18) is decomposed into two elements: the first element is called 'current cash flow' and the second element is called 'present value of future cash flow changes', as it discounts changes in earnings between accounting periods. There appears to be a subtle difference between the resource rent R_t and the current cash flow R_t^{si} (hence the additional superscript) due to a different treatment of investment goods and their depreciation. In national accounts, expenditures are either current (labor) or capital (investment goods). The resource rent therefore does not subtract the price of investment goods from output, but rather their annualized user costs (which includes a depreciation charge and an opportunity cost for the money tied up in the assets). By contrast, the current cash flow concept of sectoral income theory seems closer to business accounting, as it subtracts both current and capital costs when they are made. It can be decomposed as:

$$R_t^{si} = p_t^{mkt}(q_t - i_t) \equiv p_t^{mkt} q_t^{in,out} \tag{19}$$

Eq. 19 expresses that the current cash flows are written as the difference between commodity outflows (e.g. current production multiplied by the market price) and commodity inflows (e.g. amounts of labor, intermediate goods and investment goods i multiplied by their market prices).[12)]

[12)] Sectoral income theory in fact requires real prices (partial derivatives of the utility function with respect to consumption of commodities), however assumes in practice that these prices can be represented by market prices.

Herewith, the second element of Eq. 18 can be further decomposed as:

$$r_t V_t = R_t^{si} + \sum_{i=t}^{\infty} \frac{1}{(1+r)^{i-t+1}} [p_{i+1}^{mkt} q_{i+1}^{in,out} - p_i^{mkt} q_i^{in,out}]$$
$$= R_t^{si} + \sum_{i=t}^{\infty} \frac{1}{(1+r)^{i-t+1}} [p_{i+1}^{mkt} (q_{i+1}^{in,out} - q_i^{in,out}) + (p_{i+1}^{mkt} - p_i^{mkt}) q_i^{in,out}] \quad (20)$$

We now derive the main result of Wei (2009, p.89) where sectoral income is decomposed into three elements: current cash flow; net investment; and "price change" effects. The net investment or depletion estimate therefore becomes:

$$D_t^{si} = \sum_{i=t}^{\infty} \frac{1}{(1+r)^{i-t+1}} [p_{i+1}^{mkt} (q_{i+1}^{in,out} - q_i^{in,out})] \quad (21)$$

In order to evaluate D_t^{si} properly, one would require a specific forecast when investments are going to be made, for instance due to the expected retirement of fixed capital, new explorations and/or discoveries. In addition, one would require assumptions about the future price path of such investment goods for all companies involved in extraction.

When we compare this approach with the theoretical model of Section 3.2, it can be seen that depletion as net investment is an exemplification of Eq. 5, depletion defined as a change in value.

Expectations

Before we will test these four approaches in Section 3.5 on real data, we need to discuss the issue of expectations. This may not be necessary when working in deterministic settings as most theoretical models assume, but becomes important when working in discrete time and under uncertainty as is the case in most empirical evaluations. E.g. is the resource rent paid at the end of the accounting period or the beginning? Are the asset values evaluated before or after discoveries may have occurred? Expectations are also important for understanding the treatment of capital gains in the SNA as we will discuss in Section 3.6.

To make this explicit, following Hill and Hill (1999, 2003), we introduce an expectation operator E_s which evaluates an expression at time s, given information available at s. What are the implications for our four depletion approaches?

Regarding the theoretical depletion measures of net savings and net investments, these are both formulated in a deterministic setting, in which there is no uncertainty i.e. $E_{t+1}(V_t) = E_t(V_t)$.

Expectations are also not important for our second depletion measure (Eq. 12) as the prices and stocks are always evaluated based upon current expectations and the depletion measure is therefore unambiguous i.e.

$$D_t^{phys} = 1/2\left\{(E_{t+1}p_{t+1}^{gr} + E_t p_t^{gr})(E_{t+1}S_{t+1} - E_t S_t)\right\} \tag{22}$$

However, for depletion as change in total wealth, the depletion measure is dependent upon the moment expectations are formed. Following Hill and Hill (1999; 2003), we can formally introduce two different measures of depletion: *ex ante* and *ex post* as:

$$D_t^{ante} \equiv E_t D_t \equiv E_t V_t - E_t V_{t+1} \tag{23}$$

$$D_t^{post} \equiv E_{t+1} D_t \equiv E_{t+1} V_t - E_{t+1} V_{t+1} \tag{24}$$

In accounting, one often compiles a time series of stock accounts, which describe the changes in value over time, due to discoveries, revaluation, reclassifications and extractions/depletion. These series are consolidated, which means that the decomposition in changes matches the differences between opening and closing stocks exactly. With the help of expectation operators, we first write the opening stock as $E_t V_t$ i.e. the value at time t expected at time t, and likewise the closing stock can be written as $E_{t+1}V_{t+1}$ the value at time $t+1$ expected at time $t+1$. This reflects the fact that we always evaluate stocks based on current knowledge about prices and extraction path. With the help of Eqs. 23 and 24, we can now decompose the difference between opening and closing stocks as:

$$E_{t+1}V_{t+1} - E_t V_t = U_{t+1} + D_t^{ante} \tag{25}$$

where $U_{t+1} = E_{t+1}V_{t+1} - E_t V_{t+1}$

i.e. the unexpected capital gains – given the information available at time t – that occurred during the accounting period, evaluated with respect to the closing stock.[13] The intuition here is that if a change occurred in the value of an asset (estimated as NPV) evaluated at two different points in time this is due to unexpected capital gains (e.g. discoveries). Likewise, an alternative decomposition of the stock account can be obtained as:

[13] Hill and Hill (2003) use a slightly different notation for the unexpected capital gains.

$$E_{t+1}V_{t+1} - E_tV_t = D_t^{post} + U_t \tag{26}$$

We can now see that Eq. 25 is consistent with an *ex ante* notion of depletion, in which depletion is calculated based on expectations at the beginning of the accounting period of prices and extraction path, while Eq. 26 is consistent with an *ex post* notion of depletion in which depletion is calculated with respect to expectations as they exist at the end of the accounting period. These expectations may have changed due to changes in prices, revaluations or discoveries. The *ex ante* and *ex post* decompositions are both consistent with depletion as a change in total wealth.

With respect to Table 3.2, the *ex ante* and *ex post* depletion measures would be consistent with the SEEA CF recording proposals. The BEA/Repetto proposal would take some of the unexpected capital gains into account (the discoveries) in order to calculate the charge against production (which was shown in Table 3.2 as A–D).

To illustrate the usefulness of expectation operators, we are now in a position to clarify that what Wei (2009) calls price change effects are necessarily expected price changes i.e. $E_t(p_{t+1}^{mkt} - p_t^{mkt})q_t^{in,out}$ which is different from the notion of revaluation in national accounts, which can be expressed as $(E_{t+1}p_t^{rr} - E_tp_t^{rr})E_tq_t$. Whereas the latter describes changes in expectations for the path of unit resource rents i.e. intratemporal changes, the former element describes intertemporal price changes that are already expected. In empirical evaluations the default assumption is that the unit resource rent p_t^{rr} remains constant. Wei's 'price change effect' would herewith become 0. Moreover, sectoral income theory requires a specific path of future investment, which statisticians are often reluctant to make. The forecast one usually makes is contained in the assumption that the unit resource rent remains constant, which implies that one expects the user costs of capital per unit of extracted resource to remain constant. This implies that in most empirical applications, D_t^{si} will be equal to the *ex ante* depletion costs D_t^{ante} which is also the assumption we will make in Section 3.5.

3.5 Testing the approaches on Dutch gas reserves

In this Section we will evaluate the proposed depletion estimates D_t^{si}, D_t^{post}, D_t^{phys}, D_t^{cw} for real Dutch data on stocks of natural gas reserves between 1990 and 2010.

As we have seen in Section 3.4, all measures require the estimation of V_t first. Statistics Netherlands compiles stock accounts annually for natural gas in both physical and monetary terms, which are included in the balance sheets of the national accounts (Statistics Netherlands, 2011a).

Asset accounts in monetary terms

Estimating the stock value of natural gas V_t based upon a NPV model requires several assumptions. We follow the methodology and choices as described in Veldhuizen et al. (2009). Following Veldhuizen et al. (2009, p.8–10), some of the key choices are:

- The scope of reserves is restricted to so-called expected reserves which correspond to proven and probable reserves.
- The extraction path assumes a linearly decreasing rate of extraction until exhaustion.[14] This is consistent with the prognosis by the Dutch Government, which is based upon both company strategies as well as Government imposed production restrictions. For instance, the supply from the largest Dutch field (Groningen) is restricted by Government (for the time period 2011–2020) in order to fulfill a balance function and allow for a 'small fields policy' in which with fiscal incentives smaller fields are depleted as much as possible.[15] The extraction path is smoothed using a 3-year moving average, to account for the effects of weather extremes.
- The resource rent is obtained by deducting the user cost of produced capital from the gross operating surplus. The Dutch national accounts provide data on the gross operating surplus of the 'extraction of crude petroleum and natural gas' industry.
- The resource rent is split into an oil and gas component based upon the ratio of oil and gas production values. Given that oil production is small compared to natural gas production, we are able to identify the resource rent that represents the returns from natural gas only, quite accurately.
- The user cost of produced capital consist of three components: (1) revaluation of assets, (2) the opportunity cost of money tied up in the assets, and (3) the net taxes (taxes-less-subsidies) paid to the government. In practice the user

[14] Before 2001, a constant extraction rate is used, based on governmental policies that existed during this period in which a strict production boundary was set.

[15] There has been some debate regarding how to estimate the life length of the resource. Harrison (1997) for instance has argued that instead of dividing the physical stock by current period's extraction to estimate the life length, it would be better to divide the stock by the current period's extraction net of discoveries. However, this would go against our specific government regulated path, and Harrison's proposal could also imply infinite lifetime when discoveries exceed extraction in a given year.

costs are estimated using an exogenous rate of return which consists of the sum of the internal reference rate between banks and a risk premium of 1.5%.
- Because of the volatility of gas prices a 3-year moving average is used to estimate the expected unit resource rent.
- The real discount rate used is 4%, which equals the real discount rate used for the measurement of fixed capital in the Netherlands.

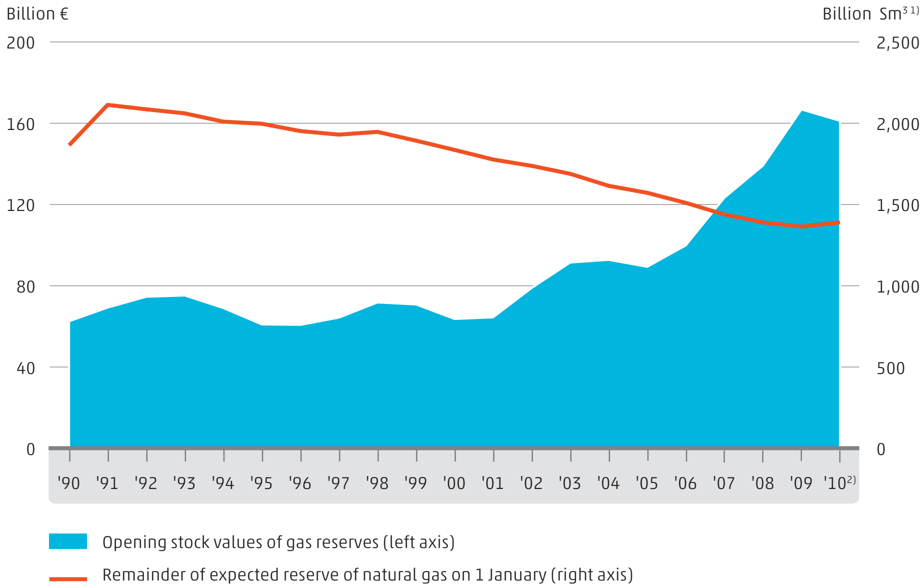
These choices ensure that the chosen extraction path and discount rates are consistent with Dutch policy. Obviously, it also implies that the extraction path is exogenous, and not influenced by the development of prices.

The results, shown in Figure 3.3, indicate that the physical stock of reserves has decreased since 1991. The monetary value peaked in 2009 at 167 billion €, but has decreased since, due to both extractions as well as price decreases.[16] Although new reserves are discovered occasionally, the cumulated production of natural gas in physical terms has exceeded remaining reserves as known today, in 1999. More than two thirds of the initial gas reserves, to current knowledge, have been depleted already due to the yearly extractions (Statistics Netherlands, 2011b).

For a correct interpretation, it is important to stress that Figure 3.3 depicts monetary estimates in current prices that is including the effect of price changes. Secondly, the value for a specific year is estimated based upon the information available at that time. Hence, stock values of year *t* are not re-evaluated based on information that has become available in later years.

[16] Obviously, the valuation is sensitive to the assumption used: when a 3% discount rate would be used, the value would increase for the considered period on average by about 12%. When a fixed extraction rate is used the value would increase on average by 10%.

Figure 3.3 Dutch natural gas reserves (1990-2010)



Source: Statistics Netherlands (2011b).

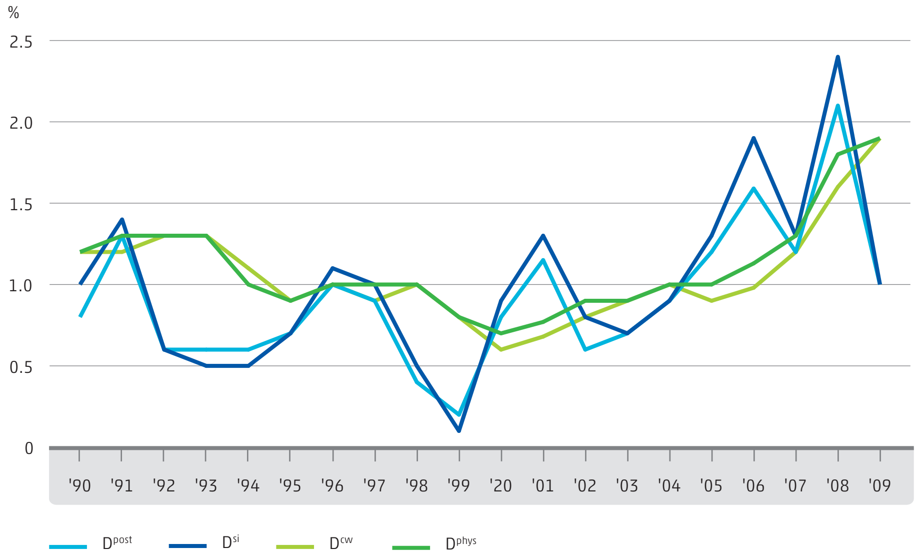
1) The 'standard' cubic meter (Sm^3) indicates a cubic metre of natural gas or oil under standard conditions corresponding with a temperature of 15 °C and a pressure of 101,325 kPa.

2) Provisional estimate.

Depletion measures

Figure 3.3 depicts the results of the calculated depletion measures as defined in Section 3.4, expressed as a percentage of net national income. For these calculations, we used the stock estimates as presented in Figure 3.3. The general picture that emerges is that depletion was lowest in 1999 and peaked in 2008. Correcting for the cost of depletion would lead to a significant downward adjustment of the level of Dutch NNI for instance of between 1.6% and 2.4% in 2008. The wide range of estimates indicates the importance of selecting a particular measure, when the goal is to obtain a depletion adjusted macroeconomic indicator.

Figure 3.4 Depletion estimates for main approaches (% of Dutch net national income; 1990-2010)



Source: Author's calculations..

When we compare the individual measures we observe that the sectoral income/ *ex ante* and *ex post* estimate follow a similar pattern. It also shows that in practice, the difference between D_t^{ante} and D_t^{post} is not very significant. However, it should be pointed out that this is primarily because no significant discoveries have been made in the past twenty years in the Netherlands. In case of large discoveries, D_t^{ante} and D_t^{post} would behave very differently.

The physical estimate is less volatile than the *ex post* and *ex ante* measure, which is to be expected as the proposed method is based upon average prices (e.g. a 3-year moving average is used to estimate the expected unit resource rent which is used for estimating the stock value). Its pattern is different from the monetary measures, which respond directly to the current period's resource rent. We observe that the comprehensive wealth and the physical estimate are obviously very close, as the latter essentially averages the asset prices of the opening and closing stock for each year.[17)]

[17)] In order to calculate the comprehensive wealth estimate, we face the difficulty that according to its underlying model a fixed extraction rate has to be used, which would compromise the comparison with the other measures which are based upon stock estimates that use a linearly decreasing extraction rate. The comprehensive wealth estimate shown here is based upon the Dutch stock value and using the current period's extraction rate, while in Section 3.6 we show the results when a constant extraction rate is assumed (as in World Bank, 2011).

Consolidated stock accounts

To illustrate how the depletion measures relate to stock estimates, Table 3.5 shows consolidated monetary stock accounts according to two possible decompositions: *ex post* depletion (Eq. 23), and the physical measure (Eq. 12). It can be seen that both decompositions are exact: the changes due to depletion, discoveries etc. add up to the difference between opening and closing stocks as evaluated with a net present value method.

Table 3.5 Monetary stock accounts for Dutch natural gas reserves in two different decompositions[1] (2000-2009)

	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009
	million €									
Opening stock	63,624	64,444	78,894	91,418	92,747	89,317	99,846	123,328	139,092	166,749
Revaluation	6,362	18,640	13,782	3,340	-2,806	14,005	29,817	21,696	29,712	-10,214
Discoveries/reclass.	-2,533	255	1,278	786	3,228	1,706	1,240	-115	8,437	9,306
Ex post depletion	-3,010	-4,445	-2,536	-2,796	-3,852	-5,182	-7,575	-5,817	-10,491	-4,660
Closing stock	64,444	78,894	91,418	92,747	89,317	99,846	123,328	139,092	166,749	161,182
Opening stock	63,624	64,444	78,894	91,418	92,747	89,317	99,846	123,328	139,092	166,749
Revaluation	2,911	16,043	14,962	5,457	-974	14,341	28,872	20,315	30,547	-8,664
Discoveries/reclass.	312	1,358	1,107	-292	1,973	683	-20	1,794	5,998	11,879
Physical depletion	-2,404	-2,951	-3,545	-3,836	-4,430	-4,494	-5,370	-6,345	-8,888	-8,782
Closing stock	64,444	78,894	91,418	92,747	89,317	99,846	123,328	139,092	166,749	161,182

[1] Ex post depletion measure (above); physical measure (below).

Effect on growth rate

We have already seen that the effect on the level of NNI is significant. However, it is often not the level of production or income that is policy relevant, but its growth. Therefore, Table 3.6 compares the unadjusted annual growth rate in net national income in current prices (obtained from the Dutch national accounts), with the various depletion adjusted growth rates that we obtain after correcting net national income for cost of depletion. The data presented here therefore are not adjusted for inflation.[18]

We see that the effect on growth would be considerable for most years depending on the chosen measure, ranging between a –1.1% adjustment towards a 1.4%

[18] The reason we present nominal growth rates is that it is not obvious for each measure how to deflate.

adjustment of the growth rate in current prices. The physical and comprehensive wealth measures result in the lowest average adjustments – in absolute terms – to growth rates, as they are less volatile.

Table 3.6 Depletion adjusted growth rates (nominal) of Dutch net national income for different approaches (2000-2009)

	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	Average difference
	%										
Nominal growth	9.5	5.4	3.8	2.7	4.8	2.2	8.2	4.8	−0.6	−5.5	
Adjusted for depletion											
Physical	9.6	5.2	3.7	2.7	4.7	2.2	8.1	4.7	−1.1	−5.6	*0.14*
Sectoral income/ex ante	8.6	4.8	4.4	2.8	4.7	1.7	7.6	5.4	−1.6	−4.2	*0.62*
Ex post	8.8	5.0	4.4	2.7	4.6	1.9	7.7	5.3	−1.5	−4.5	*0.51*
Comprehensive wealth	9.6	5.3	3.7	2.6	4.7	2.3	8.1	4.6	−1.0	−5.8	*0.17*
Largest deviation	−0.8	−0.5	0.6	−0.1	−0.2	−0.4	−0.6	0.5	−1.1	1.4	

Now that we have evaluated all depletion measures, and seen that the choice for one over another has important ramifications for both level and growth of macro-economic indicators, we turn to discussing which measure is most appropriate in the context of national accounting.

3.6 Discussion

Empirical measures of depletion, in particular the change in total wealth measure, have been criticized from time to time by theorists. Here we briefly discuss the criticisms by comprehensive wealth and sectoral income theory.

Hamilton and Ruta (2009, p.64) claim that "*standard practices in the natural resource accounting literature (particularly valuing resource depletion according to the El Serafy formula) are measuring net saving, and therefore the change in social welfare, with an upward bias*". In fact, we have according to Eq. 17 that the change in the stock value can be written as a sum of two terms:

$$\frac{dV}{dt} = \frac{d(p^{sh}S)}{dt} = p^{sh}\frac{dS}{dt} + S\frac{dp^{sh}}{dt} \qquad (27)$$

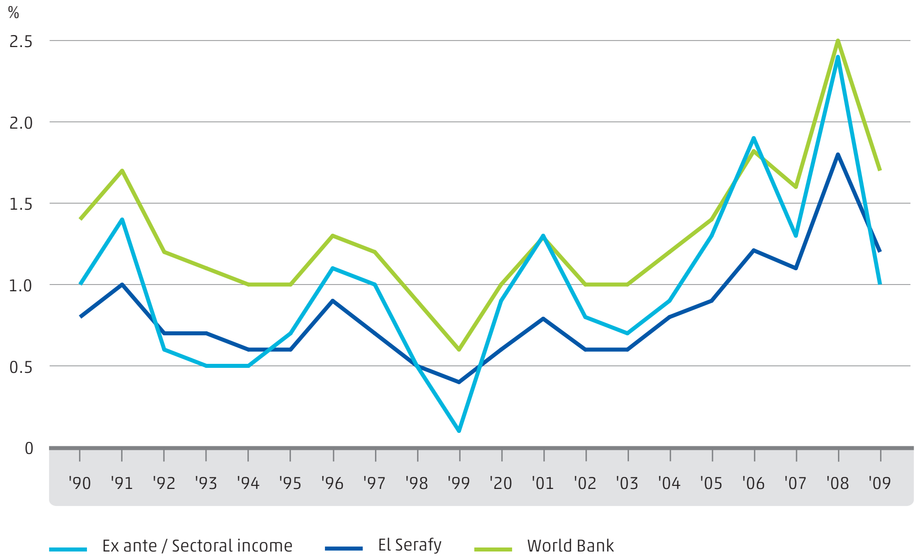
Hamilton and Ruta indeed prove that (within the context of their constant extraction model) depletion as net saving is larger (i.e. more negative) than depletion as change in total wealth i.e. $p^{sh}\dot{S} > \dot{V}$. Empirical applications would therefore underestimate depletion (i.e. their depletion measure would be less negative) and therefore overestimate the change in social welfare.

To corroborate these claims, Figure 3.7 compares D_t^{ante} – as an example of a change in total wealth measure – with the World Bank's constant extraction depletion measure.[19] Although the World Bank estimate [assuming constant extraction rate until exhaustion; no smoothing of resource rents] is larger (i.e. more negative) than D_t^{ante} for most years, this is not true for all years. For instance, the D_t^{ante} estimate is in 2001 and in 2006 slightly higher (i.e. more negative) than the World Bank estimate. This is because of the large difference in stock value estimates (about 80% in 2001) caused by the choice of a constant extraction instead of a linearly decreasing path. A lower stock value commands a lower return, and hence according to Eq. 8 results in higher depletion estimates. Figure 3.7 also includes an *ex ante* depletion estimate based upon a constant extraction rate (the so-called El Serafy approach), in which we assumed the current period's resource rent constant until time of exhaustion. We see that the El Serafy estimate is indeed smaller than the World Bank estimate. Therefore, we conclude that the validity of the criticism depends on what one considers 'standard practice'. The criticism is no longer valid when we move beyond a constant extraction scenario.

Wei (2009, p.89) also criticizes empirical accounting: "*The change of sectoral wealth is associated with the value of depletion of resources in the practice of non-renewable resource accounting (ref. SEEA, 2003). This view of depletion of resources may be misleading since the change of sectoral wealth may result from price change effects besides net investments*." However, based on our discussion of expectations in Section 4.5 we are able to see that this criticism seems misguided as these price changes are expected and already included in the value of the stock V_t and therefore also in income rV_t. Including these anticipated price changes in income but excluding them from depletion would be inconsistent.

[19] In practice, the World Bank (2011) defines energy depletion as the ratio of the present value of rents, discounted at 4%, to exhaustion time of the resource. The exhaustion time of the resource is defined as Min (25 years, Reserves/Production). The estimates therefore assume – as required by the model – a constant extraction rate to exhaustion.

Figure 3.7 Comparison of ex ante depletion with two constant extraction measures (% of Dutch net national income; 1990-2010)



Source: Author's calculations.

Nevertheless, what these criticisms on statistical applications demonstrate is that which depletion estimate is favored depends on the context of use. Asheim and Wei (2009) and Hamilton and Ruta (2009) are concerned with income – and a fortiori depletion – as a measure of social welfare, which according to Hamilton and Ruta (2009), is the more policy relevant measure.

However, an alternative context of use for assessing depletion is the measurement of sustainable income. The SEEA 2003 (UN et al., 2003) explicitly refers to Hicks' well known (1946) definition of income (i.e. "the maximum amount an individual can consume during a period and remain as well off at the end of the period as at the beginning") in its introductory paragraphs as a means to operationalize sustainability (para 1.18). A foundation of environmental accounting in Hicksian income has the advantage as shown by Hamilton and Ruta (2009) that it leads to an interpretation of income as the return to wealth. Within such a context of use, measures of depletion defined as a change in total wealth would be the preferred choice.

In particular, based on the arguments put forward by Hill and Hill (1999; 2003) it can be seen that D_t^{post} would be preferred over D_t^{ante}. The basic intuition is that windfall gains (e.g. gas discoveries) could be consumed straightaway and technically would make us still as well off at the end of the period as at the beginning, however such a level of income would not be sustainable. Hill and Hill (2003) therefore demonstrate that the sustainable income measure should be based upon revised expectations at the end of the accounting period. This would be consistent with *ex post* depletion. By contrast, the *ex ante* depletion measure would not take these discoveries into account at all.

A third context of use that we may distinguish is pragmatic. When depletion is used to adjust NDP, the argument goes, one primarily needs to ensure that it is fully consistent with the income concept that underlies the SNA. This is easier said than done, as there is a lot of controversy regarding the income concept that is and or should be used in the SNA.[20] A large part of the controversy has to do with the treatment of so-called capital gains or holding gains i.e. changes in value due to changes in the structure of prices that occur during the accounting period.[21] The SNA is strict in ruling that holding gains are not considered to form part of income, as they are not transactions.[22]

Now, a basic characteristic of using a NPV model is that the value of the original deposit automatically increases (in the absence of discoveries/reclassifications etc..) each year by 'the effect of time passing' which is called in the 2008 SNA 'the unwinding of the discount rate' due to the fact that future income streams are discounted one period less. To see this, by revisiting Eqs. 5 and 6, we find that in case of a non-stationary deterministic economy (see also Cairns (2009, p.124)) the following holds:

$$rV(t) = \pi(q(t)) - \lambda q(t) + \frac{\partial Z}{\partial t} \tag{28}$$

In the absence of extraction during the accounting period[23], for instance due to Government demanded suspension of operations, the return to the asset rV_t equals the unwinding of the discount rate. It also demonstrates that in such instances (given the validity of Eq. 8) we may end up with recording positive depletion (as the resource rent would be zero) which according to some scholars is unsatisfactory.

[20] Item 3 on the long term research agenda for the SNA is "Clarification of income concept in the SNA - should holding gains be included" (unstats.un.org/unsd/)

[21] "Holding gains are sometimes described as "capital gains", but "holding gain" is preferred here because it emphasizes that holding gains accrue purely as a result of holding assets or liabilities over time without transforming them in any way." 2008 SNA, para 3.105.

[22] This stance has been criticized from time to time of being out of touch with reality. To give an example, suppose a house increases in value during the accounting period. This constitutes a holding gain, but as no transaction occurs, does not form part of income, although some households may use the increased value to take out an extra loan and spend the money freely. In fact many theorists (e.g. Hartwick, 2002) argue that capital gains (which would be expected as in case of future price changes) should be included "in order for net national income to be correct".

[23] This example was brought forward in Gravgård (2010).

Now, the SEEA Central Framework (UN et al., 2012, p. 206) shows that the depletion adjusted resource rent (or net income) according to the physical depletion measure can be written as rV_t less (expected) revaluation of the asset.[24] It is argued that as r is a forward looking rate it includes expected holding gains, the expected revaluation therefore has to be subtracted in order to obtain an income measure devoid of (expected) holding gains.[25] Therefore, D_t^{phys} would be fully consistent with the SNA income concept.[26]

As a final observation, when using the physical approach to correct national income, the assumptions used in valuing stocks should be considered carefully. For example, use of price smoothing in estimating expected future resource rents, may result – e.g. in case of a rapid price decrease during the accounting period – in a depletion estimate which would be larger than the actual resource rent as the effect of the price decrease on price of the asset in the ground is lagged (a situation of 'negative income').

The SEEA Central Framework has changed its initial support for the change in wealth approach towards the physical measure of depletion. The motivation for this evolution seems to be that the core of environmental accounting is about integrating physical and monetary data. Similar to the physical flow accounts that were discussed in Chapter 2 – one of the key building blocks of SEEA – which allow deriving hybrid indicators such as water productivity or energy intensity, a tight integration is obtained between stock accounts in physical and monetary terms. The SEEA Central Framework has tried to formulate a depletion measure that is as much aligned with physical stock accounts as possible, while still respecting the accounting rules of the SNA. Although there are clear advantages to this position, the choice for the physical depletion measure also comes at a cost: we forego a foundation of environmental accounting in Hicksian income or social welfare.

[24] This is shown (UN et al., 2012, p. 205-206) in an ex ante perspective i.e. ignoring discoveries etc. Formally, in the notation of this chapter we would have $E_t(R_t - D_t^{phys}) = E_t(rV_t - 1/2(S_{t+1} + S_t)(p_{t+1}^{gr} - p_t^{gr}))$. To give a simple example: suppose we have at t0 a physical stock S of 50 worth 100, hence a price per unit of 2; assuming a discount rate of 10% the ex ante expected return to the asset rV would be 10; in the absence of discoveries and extraction, the value at the end of the accounting period would be 110, resulting in a price at the end of the period of 2.2; we therefore also have a revaluation of (2.2-2) *50 = 10; thus, the depletion would indeed be 0 as expected as there is no extraction.

[25] This view would be supported by Usher (1994, p136) who writes that "a capital gain ... may reflect ... the mere passage of time ...[which] is excluded from the measures of income in the National Accounts because of the essential this-yearness of national income."

[26] It should be mentioned here however, that there exists also an alternative interpretation of the income concept of the SNA (e.g. Hill and Hill 1999, 2003) which argues that only unexpected holding gains are excluded from the income concept of the SNA, but that expected holding gains, such as due to time passing, are included. One can find support for this position from within the 2008 SNA itself: "This increase in value, in common with the increase in the value of any asset due to the unwinding of a discount factor, is treated as income in the SNA" (UN et al., 2009, p.380).

3.7 Conclusions

We have given a review of the most important empirical and theoretical depletion measures that have recently been put forward and explained how they compare to the basic theoretical model of a firm engaged in extracting a non-renewable resource. We have explained that in practice due to empirical realities as non-optmality and uncertainty we are forced to go beyond the classic theoretical model.

We have been able to evaluate all measures in practice by testing their behavior on Dutch time series of natural gas reserves, using default assumptions in Dutch accounting. We have seen that correcting for depletion has a significant effect on both the level as well as the growth rate of net income which varies per measure. The choice for a specific measure is therefore highly policy relevant.

We argued that the choice for a depletion estimate should be determined by the context of use. When the context of use is that of measuring sustainability *ex post* depletion is the most feasible candidate. By contrast, theoretical depletion measures are more often concerned with social welfare measurement. The SEEA Central Framework favors depletion as 'using up' of the resource for pragmatic reasons, most importantly consistency with the income concept of the SNA.

We have seen that there is a lot of confusion between theory and practice about basic concepts such as 'rent' and 'price'. It is a pity that there has been little interaction between theorists and practitioners in environmental and green accounting, as they have each other much to offer. Practitioners could obtain a better grasp of the overall context, while theorists could be served by better realizing whether their theories could be readily implemented in practice.

4. Towards a consistent approach for ecosystem accounting

4.1 Introduction[1)]

There is a still growing interest in better understanding the economic implications of the ongoing changes to the world's ecosystems (MA, 2005; TEEB 2010; EC 2011; UK NEA, 2011). Among others, there has been a strong increase in interest in developing 'ecosystem accounts', building on the experiences gained with environmental economic accounting since the mid-1970s. As ecosystem accounting is not a standardized concept, we will define it here as the integration of ecosystem services and ecosystem capital into national accounts. The increasing interest in ecosystem accounting is illustrated in, for example, the recent EU Biodiversity strategy (EC, 2011) which calls upon Member States to *"assess the state of ecosystems and their services in their national territory by 2014 and assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020"*. The progress in analyzing, modeling and valuing ecosystem services (e.g. Daily et al., 2009; De Groot et al., 2010) is facilitating the further development of ecosystem accounts. Early studies in the field tended to focus on the economic benefits provided by individual ecosystems, but there are now increasingly also studies that analyze ecosystem services at landscape, national or even continental level (see e.g. TEEB, 2010).

Developing and applying ecosystem accounting methods requires the physical and monetary measurement of (changes in) ecosystem services supply and the capacity of ecosystems to supply services to be recorded in a way that is aligned with the measurement approaches prescribed for national accounts (as reflected in the System of National Accounts; SNA) and for environmental economic accounts (as reflected in the System for Environmental-Economic Accounts Central Framework; SEEA CF). The SEEA CF is, as of 12 February 2012, a global statistical standard for environmental accounting (UN et al., 2012). However, neither the SNA nor the SEEA CF were designed for accounting for ecosystem services or ecological capital. For instance, the compartmental approach to natural resources applied in the SNA and SEEA CF is not easily aligned with the ecosystem service concept and the notion of ecosystems being a functional unit delivering multiple services to multiple stakeholders (e.g. Hein et al., 2006). To date, therefore, there is still insufficient understanding of how ecosystem services, once quantified, can be incorporated in an accounting framework such as the SNA or the SEEA (Mäler et al., 2009; Campos and

[1)] This chapter is based upon (with only minor changes) Edens and Hein (2013b). The authors would like to thank Carl Obst, Alessandra Alfieri, Ivo Havinga, Mark de Haan and three anonymous references for their suggestions and comments. In addition, the authors gratefully acknowledge Carl Obst for his suggestions regarding the allocation of ecosystem services to institutional sectors. The authors also would like to thank members of the Editorial Board of SEEA EEA as well as participants in the Expert Meetings on Ecosystem Accounting in Copenhagen and London for useful discussions and insights.

Caparrós, 2011; Banzhaf and Boyd, 2012). In recognition of these issues, the SEEA Experimental Ecosystem Accounting guidelines have recently been developed by a consortium coordinated by the United Nations Statistics Division (UNSD, 2013); both authors have contributed to these guidelines.

The specific objective of this paper is to identify and analyze key methodological challenges related to the construction of ecosystem accounts. In particular, we review the efforts undertaken to date to incorporate ecosystem services into national and environmental accounts, identify four key challenges to be addressed when guidelines and potential approaches for ecosystem accounting are put in practice, and provide a number of specific recommendations and potential ways forward. These four issues were also recognized as needing further research in the SEEA EEA (UNSD, 2013). Testing and implementation of the SEEA EEA is the responsibility of national statistical offices together with a range of other agencies, and will involve the detailed biophysical and monetary quantification of ecosystem services and ecosystem capital. Some of the key issues that need to be resolved when the SEEA EEA guidelines are applied are addressed in this paper, and we hope this paper will contribute to the broader scientific debate on ecosystem accounting as well as provide a number of specific recommendations for the actual implementation of ecosystem accounts.

The set-up of the paper is as follows. First, we present a brief introduction to the complex topic of environmental and ecosystem accounting, in the context of the SNA, briefly highlighting the main developments in this field since the mid-1970s. Second, we analyze four key challenges in the field of ecosystem accounting, examine how these challenges have been addressed in the accounting and ecological economics literature, and present a consistent, conceptual approach to address these challenges. Third, in the Discussion Section, we place our findings in the context of the ongoing efforts to develop guidelines and methods for ecosystem accounting. We present our key outcomes in the Conclusion Section. In an (on-line) appendix we present an illustration of how ecosystem services and ecosystem capital can be incorporated in a satellite sequence of accounts.

4.2 The development of ecosystem accounting

The accounting context

As discussed in Chapter 2, while the SEEA CF provides a much broader perspective on the environment than the SNA, it does not provide an analysis of ecosystem services or ecosystem capital. One of the main reasons is that while the SEEA CF relaxes the asset boundary, it keeps the SNA production boundary intact. For produced assets, the production boundary constrains the asset boundary, but this does not apply to many natural resources which are considered non-produced assets i.e. they are not the outcome of production processes and the services they provide are considered rent payments. Consequently, both the SNA and SEEA exclude from the production account various types of ecosystem services such as regulating services as well as the natural growth of biological assets. In addition, while the SEEA CF provides recommendations on the treatment of depletion, it does not contain a discussion of the treatment of environmental degradation or rehabilitation.

The Convention on Biological Diversity defines an ecosystem as 'a dynamic complex of plant, animal and microorganism communities and the nonliving environment, interacting as a functional unit' (United Nations, 1992). Importantly, ecosystem dynamics and the supply of ecosystem services depend on the functioning of the ecosystem as a whole, rather than on specific components in isolation (e.g. Potter et al., 1993; Arshad and Martin, 2002; Van Oudenhoven et al., 2012). One of the challenges of ecosystem accounting is to integrate the complex and multi-facetted concept of the ecosystem with the compartmental approach of the SNA accounting structure. Furthermore, in an ecosystem approach, the distinction between cultivated and natural assets is difficult to make; there are few if any ecosystems left on the planet that are not strongly modified by people, and even in cultivated assets ecosystem dynamics and natural processes remain important.

The different environmental and ecosystem accounting approaches

It is not possible to provide an overview here of the vast environmental and green accounting literature (but see Heal and Kriström, 2005 for an overview). Instead, in this section we will zoom in on a number of contributions pertinent to ecosystem accounting. As we have seen in Chapters 2 and 3, national accounts focus not on welfare but on measures of economic activity as defined by SNA system boundaries. For example, the measure of value created in national accounts (GDP) is more aligned with the producer surplus concept than with a total welfare measure. There are also numerous studies that aim to develop indicators for social welfare (e.g. Nordhaus and Tobin, 1972; Daly and Cobb, 1989), but as we focus on integrating ecosystem services in a strict national accounting framework we do not consider them further in this chapter.

Henry Peskin (1976) discusses already in general terms how values of service flows and damages associated with the use of environmental assets can be integrated into the national accounts. In the context of an input-output table he introduces an additional 'natural sector' which produces what he calls 'environmental asset services and net environmental benefits' and consumes environmental damages. Peskin provides four alternative adjustments to what was then called gross national product and concludes that there is 'no best way' as the feasibility of each approach depends on the context of use: i.e. in support of environmental management; measurement of welfare; or establishing an index of productive services.

During the 1980s, several countries started compiling environmental accounts which led to the development of the first handbook for SEEA in 1993 (UN, 1993). The SEEA 1993 discusses accounting for ecosystem services as one of several possible extensions to a core set of accounts, in particular as version V.2 (UN, 1993). Three types of services are distinguished: disposal services, productive services of land (e.g. use of soil for agricultural purposes), and consumer services (e.g. amenity services). However, the description of these services was limited to a recording of the decrease of these services and did not recognize services in a productive sense. The reason being that the focus of SEEA 1993 lies on adjusting macro-economic aggregates for the cost of natural resource depletion and environmental degradation, which came to be known as estimating a 'green GDP'. It proposed to estimate these costs based upon the maintenance costs approach, which consist of the hypothetical costs required to restore the environment to a previous state. Peskin and Delos Angeles (2001) propose an alternative accounting framework with the acronym ENRAP which was developed during the Environment and Resource Accounting Project carried out in the Philippines in the 1990ies. Most importantly, they criticize SEEA 1993 for only accounting for the depreciation of natural resources (e.g. depletion) but not for its positive non-market outputs (termed environmental quality services), which they propose would add to GDP.

A very different approach was followed in the research by the World Bank on genuine savings (Hamilton, 1996; Hamilton and Clemens, 1999). The point of departure is to use macroeconomic models to analyze issues such as pollution, resource extraction, pollution abatement costs as well as ecosystem service supply, using optimal control techniques. Ahlroth (2000), Skånberg (2001), and Gren (2003) follow a similar approach in estimating 'green GDP' type of measures based on theoretical models in which both degradation as well as ecosystem services are included. The valuation methods they proposed reject the maintenance cost approach in favor of using benefit and damage based pricing techniques.

Eurostat (2002b) also makes an important contribution to the field. It discusses the incorporation of environmental and recreational functions in a forest accounts framework (the framework is called Integrated Environmental and Economic Accounting for Forests (IEEAF); Eurostat, 2002a) based on results of several pilot studies (e.g. Statistics Sweden, 2001) including an enumeration of double counting issues. However due to several theoretical and practical issues inherent in integrating these values into an accounting system it was decided not to include these services in the IEEAF.

Given these conflicting approaches towards valuation, the successor framework of the SEEA 1993, the SEEA 2003 (UN et al., 2003) included a separate section on land and ecosystem accounts, but limited the description to accounts in physical terms. On the valuation of depletion and degradation, the SEEA 2003 refrained from providing unique recommendations, and instead resorted to providing multiple options, which is one of the reasons why it fell short of being a statistical standard. The SEEA 2003 also does not contain a systematic discussion of ecosystem services. Many countries – especially in the European context – hereafter shifted their focus from accounting in monetary terms towards compiling physical accounts, given the difficulties around valuation and double counting. Several country-level studies, for instance in Germany and Sweden, estimating 'green GDP' types of measure achieved mixed results in producing monetary estimates of changes in environmental variables (Hecht, 2001).

The topic of ecosystem accounting continues to attract interest from researchers and policy makers. Boyd and Banzhaf (2007) have drawn attention to the importance of defining ecosystem services in such a way as to make them comparable to conventional accounting notions, such as distinguishing between price and volume. In a later contribution, Banzhaf and Boyd (2012) formulate the issue more generally as an 'index number problem' and they propose an Ecosystem Services Index which would be on par with GDP rather than adjust GDP. Mäler et al. (2009) discuss accounting for ecosystem services in the context of a wealth based accounting system. They argue that the key challenge is not so much valuing ecosystem services as such, but rather to estimate the correct accounting prices (i.e. the contribution to social welfare) of changes in stocks. Based on a number of examples they demonstrate that the accounting prices are dependent on how the dynamics of a particular ecosystem are modeled. The World Bank (2011) recognizes that while some of the regulating ecosystem services are likely to be reflected already in the value of agricultural land, amenity services associated with other types of land are not commonly included in this value. The World Bank continues work in this field in, among others, the ongoing WAVES (Wealth Accounting and the Valuation of Ecosystem Services) Project.

4.3 Key challenges in ecosystem accounting

Defining ecosystem services in the context of accounting

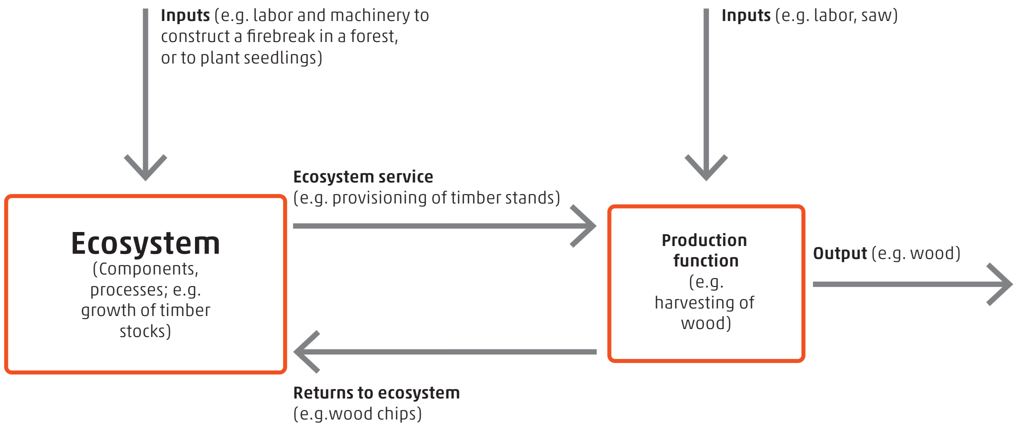
Key issues. The first of the challenges is to come up with a consistent definition for an ecosystem service, that can be applied in an accounting context. Various definitions of ecosystem services have been provided in recent contributions (MA, 2003; Boyd and Banzhaf, 2007; TEEB, 2010; Bateman et al., 2010). A key issue is if ecosystem services are *the benefits* provided by ecosystems (e.g. MA, 2003), or *contributions* to these *benefits* (e.g. TEEB, 2010). In the case of accounting, there is a need to very specifically define what an ecosystem service is and how this service is generated as a function of ecosystem activity and other inputs (e.g. labor, capital goods). A second issue is that it needs to be recognized that the large majority of ecosystems have been modified by people, often with the specific aim of increasing the supply of specific outputs, as in the case of the conversion of forests to crop land. For instance, in natural parks, hiking trails may have been constructed in order to disclose the scenery to visitors, and firebreaks may have been constructed in a forest to control fire risks.

Proposals. To address these challenges, we propose the framework presented in Figure 4.1 below. Note that our framework implies that the distinction between cultivated and natural biological resources of the SNA ceases to exist. It is explicit that the large majority of ecosystems on the planet are to a higher or lower degree modified by people (MA, 2005). In an accounting context, the costs incurred in the past to modify the ecosystem, or the benefits obtained from these modifications, are 'sunk costs', they are reflected in the current state and value of the ecosystem, and thereby in the current capacity of the ecosystem to provide services (both the type of services and the productivity).

We propose to define ecosystem services, in the context of ecosystem accounting, as the contributions of ecosystems to productive activities (such as timber harvest) or to consumptive activities (such as enjoying the recreational opportunities offered by an ecosystem). In the case of provisioning services, the contribution of the ecosystem (for instance a standing stock of timber) is used as input into a production process (e.g. logging, which also requires the use of labor and produced capital). Provisioning services represent the final output from the ecosystem (a physical flow), as used in a productive activity (e.g. Ansink et al., 2008). In the overall chain of economic activities in which ecosystem outputs are used, it is not always easy to pinpoint the ecosystem service. For instance in the case of dairy production, cattle may feed on grass from a nearby pasture but typically also benefit from additional feed, veterinary care, shelter provided by the farmer, milking, etc. Hence, in this case the physical output (flow) that is most closely associated with the ecosystem is not the produced milk (since it depends on a whole range of other inputs as well), but the amount of grass eaten by the cattle. In the case of an improved pasture, grass production may in turn depend on human activities such as the sowing of high productive grass species, irrigation or drainage, weeding and fertilizer application. Hence, even in the ecosystem, natural (e.g. photosynthesis) and man-made inputs (e.g. seeds of high-productive grass species) are combined. However it is not possible to meaningfully disentangle the contributions of these separate elements to the production of grass, and many of these different elements do not comprise a flow (e.g. nutrient retention in the soils of the pasture). Hence, our proposal is to interpret the ecosystem service as the flow/output most directly connected to the ecosystem (e.g. the standing stock of timber that is harvested or the grass that is extracted from the pasture), while recognizing that this flow is, in the case of many ecosystems, the consequence of a combination of natural/ecological processes and human modifications to the ecosystem.

In the case of regulating services, there is no extraction, but the service has a beneficial, external impact on economic activities or on people. For instance, flood regulation by coastal or riparian ecosystems reduces flood risk thereby facilitating productive activities (e.g. the operations of a factory) and allowing people to live safely. Regulating services can only be understood by analyzing the scale at which they operate and the specific mechanisms through which they generate benefits. For instance, pollinators support agricultural production through local-scale activities (foraging, exchange of pollen). Economic valuation of a specific forest patch requires monetary analysis of the specific contribution of pollinators residing in this forest patch to crop production in nearby fields (e.g. Rickets et al., 2004). However at an aggregated level, as for instance in national accounting, crop production is already accounted for. In this case, pollination supported by forest ecosystems in nearby agricultural fields may be considered by attributing part of the benefits of the ecosystem service 'crops' generated in agricultural land to the ecosystem service 'pollination'; generated in nearby forest ecosystems, in order to avoid double counting (e.g. Hein et al., 2006). In practice, however, this level of refinement in ecosystem accounting will be difficult to achieve.

Figure 4.1 Ecosystem services in the context of ecosystem accounting [1)]



1) Human inputs are used (i) for modifying the ecosystem in such a way that it produces the ecosystem services required by the land manager; and (ii) for harvesting or using the ecosystem services. Note that a provisioning service is generated at the time of harvest or extraction (which in the case of wood production may be only once in several decades, for a specific forest area).

In the case of cultural services, the interaction may involve visiting the ecosystem, or enjoying its presence in a more passive manner. It is often specific attributes of the ecosystem that are relevant to the cultural services, for instance the presence of attractive views in a landscape, or specific species relevant to cultural or religious activities. Hence, the cultural services may not be strongly dependent on the ecological quality of an area, except in the case of eco-tourism. A specific issue pertains to biodiversity, or nature conservation. This aspect of an ecosystem supports the functioning of the ecosystem and it can also be seen as an output in itself, since people value species diversity or the conservation of rare or threatened species (Mace et al., 2012).

It is important to realize that many ecosystem services are already included in economic accounts following the SNA. For instance, harvested crops and the turn-over of recreation companies are already in the production boundary of the SNA. In this case, ecosystem accounting makes it explicit that these benefits are generated using services from ecosystems, and also allows insight in the total benefits generated by ecosystems and how these change over time. Even the contributions of regulating services are, to a degree, already reflected in the national accounts. For instance, in the case of flood protection or air filtration, the current economic activities, and their recording, reflect that these regulating processes are taking place. Without regulating services, for instance due to future ecosystem degradation, the level of economic activity may be lower (e.g. due to floods) – or additional mitigation measures may need to be taken (e.g. dyke construction). Several cultural services (such as visitors' enjoyment of a park) may not enter any production function but are included in the definition as contribution to consumptive activities. These services typically lie outside the SNA production boundary. Their inclusion in ecosystem accounts and subsequently in national accounts, would consequently, increase the overall GDP of a country, however care needs to be taken in the sense that consumer surplus is not included in the SNA accounting methodology, as further analyzed later on in this Section.

Allocating ecosystem services to institutional sectors

Key issues. In the 2008 SNA production is defined as "an activity carried out under the control and responsibility of an institutional unit that uses inputs of labor, capital, and goods and services to produce outputs of goods or services" (UN et al., 2009, para. 6.2). There is an important additional conditionality: "All goods and services produced as outputs must be such that they can be sold on markets or at least be capable of being provided by one unit to another, with or without charge (ibid., para 1.40)." There are therefore two conditions: (i) being under control of an institutional unit and (ii) marketability – that exclude many ecosystem services from being considered productive. Recognition of ecosystem services in an accounting framework therefore necessitates an extension of the production boundary.

An additional issue concerns the recording of environmental protection expenditures (sometimes called defensive expenditures) by government, which according to standard accounting conventions is recorded as final demand. The two alternatives most often mentioned are to reallocate these costs either as investments or as intermediate consumption of the economic activity that benefits from these expenses (Skånberg, 2001; Campos and Caparrós, 2006).

There are different approaches that might be employed to achieve an extension of the production boundary. However, there are important implications, with respect to who is considered the producer of these services, associated with these alternatives. It is useful to distinguish between two approaches: *ecosystems as assets used in production* or *ecosystems as independent producers* (UNSD (2013) refers to these approaches as Models A and B). Both are consistent with the conceptual framework introduced above.

i. *Ecosystem as assets used in production*. The first approach considers ecosystems primarily as assets which produce ecosystem services much akin to the concept of fixed capital producing capital services (e.g. a truck provides transport services in the form of ton kilometers). This approach can be formalized by introducing a production function F which uses ecosystem assets together with other assets such as produced capital and labor and other inputs (e.g. fertilizers) to produce outputs:

$$x \equiv F(i, K, L, E) \tag{1}$$

with

x outputs (SNA and non SNA);
K produced capital;
L labor;
E ecosystem capital;
i inputs.

This model assumes that the ecosystem asset is owned by one of the standard institutional units. It entails an extension of SNA production boundary by relaxing the condition of marketability. To give an example, while according to the classical SNA view, a farmer buys a piece of land and produces agricultural products, in this approach, the farmer is conceived as buying an agricultural ecosystem, which allows him to produce not only agricultural products but at the same also provides non-SNA outputs such as carbon sequestration or amenity services.

ii. *Ecosystems as independent producers.* An alternative approach is to see ecosystems as independent producers of ecosystem services similar to an autonomous establishment (say a factory).

These services are subsequently used by other activities or consumed. The interpretation of F is rather that of an ecological production function which applies only to the ecosystem. This function is determined by the dynamics of the underlying ecosystem, which may be expressed as (e.g. Banzhaf and Boyd, 2012, p.5):

$$s \equiv F(E) \tag{2}$$

with
s ecosystem services;
E ecosystem capital.

This model also entails an extension of the SNA production boundary but in a different way: it relaxes both the condition of marketability and the condition of being under control of an institutional unit. In fact, conceiving of ecosystems as independent producers implies that ecosystems are recognized as additional types of institutional units. Therefore, it naturally leads to an additional sector 'ecosystems' in addition to the standard institutional sectors in the economy, such as the household or the corporate sector. Such a proposal for a separate sector ecosystems can be found in several accounting proposals (e.g. Peskin, 1976; Harrison, 1993; Vanoli, 1995), although it is an outstanding issue whether such a sector should be seen as being part of the economy or part of a separate entity 'Nature'.

Table 4.1 Two approaches towards recording ecosystem services (fictional data)

	Ecosystem as asset		**Ecosystem as producing unit**		
	land owner	total	land owner	ecosystem	total
Production account					
Output - SNA	200		200		
Output - non-SNA	30			110	
regulating	*30*			*30*	
provisioning				*80*	
Total output	230	*230*	200	110	*310*
Intermediate consumption SNA	40		40		
Intermediate consumption non-SNA			80		
Total intermediate consumption	40	*40*	120		*120*
Gross value added	190	*190*	80	110	*190*

To illustrate that these two approaches imply different recording mechanisms, Table 4.1 contains a hypothetical example of an economy that consists of a farmer and a household. The farmer owns land and produces crops valued at 200; has an intermediate consumption of 40 of for example fuel; it assumes the existence of regulating services of 30 (e.g. carbon capture) and provisioning services (production of crops) of 80.

In the ecosystem as asset approach the land owner is credited with the additional non-SNA output of 30, and with the value of the crops (200). The provisioning service is not recorded as a non-SNA output as its contribution is already accounted for in the SNA output of the crops involved. The provisioning service may be separately identified as an 'of which' item of the gross operating surplus as shown in the Appendix. In the ecosystem as producing unit approach, the ecosystem is introduced as an additional sector and credited with output of 110 which includes both the regulating service (value 30) and the provisioning service (value 80). There is no double counting, as the provisioning service is subsequently intermediately consumed by the land owner reducing its value added compared to the standard situation. While the total gross value added of the economy would be equal in both recording schemes (190), total output would be higher in case of ecosystem as sector approach (310 > 230). Value added of the farmer is less in the producing unit approach, however as explained in the Appendix, due to a transfer, they would obtain the same income as under standard accounting conventions.

Both recording mechanisms have their strengths and weaknesses. For example, recognition of the fact that land owners produce a range of services – including non-market outputs – is the basic idea behind the concept of payment for ecosystem services. However, it may be strange to attribute the production of non-market ecosystem services to land owners when they are not aware or do not actively pursue the production of these kind of services. A complicating situation may arise when landowners and government hold a shared responsibility for the management of specific areas

On the other hand, what is counter intuitive regarding having ecosystems as autonomous producers is that statistical units are usually considered to be active and in control of the inputs required for production, which is not in line with the observation that many ecosystems on the planet are managed for the production of specific services by institutional sectors.

Proposals. What we therefore propose is to use both recording mechanisms depending on the nature of the service (in line with Campos and Caparrós, 2006). Regulating services often have a strong public good character: their benefits accrue to the whole of society and cannot be captured by a single industry, nor can they generally be owned unless markets for ecosystem services are set up, which is currently the case for instance for carbon sequestration and hydrological services in specific circumstances. Provisioning services often have a private goods character: their benefits are excludable and rival, accruing to specific economic actors.

Services of a public nature may be depicted as being produced by an ecosystem sector (ecosystem as a producing unit approach), while services of a private nature may be depicted as being produced by the economic sector which reaps the benefits (ecosystem as asset approach). In terms of compilation, a land/ecosystem account would describe the main types of ecosystems (e.g. agricultural land; forest land; wetlands etc.) juxtaposed with the main types of services that they provide. A land cover change matrix (see SEEA CF, para. 5.276) would describe changes in extent between ecosystems during the accounting period. Integration of the land account with information about land ownership (for instance with data from land registries) would allow to relate the supply of ecosystem services to institutional sector and/or economic activities as well as with the market value of land.

Many ecosystems provide a bundle of services which may include services of both nature at the same time. The value of the ecosystem as a whole should reflect all the services it provides, so when additional non-market services are brought within the production boundary (and assuming no relocation of protection expenditures), the asset value of the ecosystem increases compared to the market value of land which is included in the balance sheet of the economy. In mixed situations of both private and public services supply, a practical way forward would be to partition the ecosystem asset into a part which presents the value of private benefits which is assigned to the land owner (this would be the market value of land) and a part which represents public benefits which is assigned to the balance sheet of the ecosystems sector. A recording example is elaborated in the Appendix. Valuation is discussed in more detail later on in this Section.

Concerning the second issue of the recording of environmental protection expenditure by government as final demand; we propose not to relocate these expenditures towards intermediate consumption. This would require a change to one of the fundamental principles of national accounts, namely the boundary between what is considered intermediate and what is considered final consumption. Moreover, it would become difficult to draw the line, as there are many defensive expenditures (e.g. police, the army) that may be candidates of relocation. In our perception, such a drastic overhaul would rather constitute a rival accounting system than a satellite account.

Defining and measuring degradation in ecosystem accounts

Key issues. A third methodological challenge is how to measure and account for ecosystem degradation. Ecosystem degradation is a complex process that may involve a combination of changes in ecosystem properties and processes, such as (i) changes in biomass or species composition; (ii) a loss of net primary production (NPP); and (iii) changes in soil properties such as soil organic matter content (Lambin et al., 2001). Such changes can affect ecosystem functioning, for instance changing ecosystem resilience and/or the ecosystem's capacity to supply ecosystem services. Different perspectives on degradation tend to stress different aspects, for instance from a nature conservationist perspective changes in biodiversity/species composition and loss of resilience may be particularly relevant. The key contentious issues in recording degradation are: (i) measuring degradation in physical terms; (ii) measuring degradation in economic terms; (iii) allocating degradation to different sectors, as analyzed below.

i. *Measuring degradation in physical terms.* The relation between ecosystem condition indicators, such as soil organic matter, NPP or groundwater table, and the supply of ecosystem services will generally be complex, i.e. non-linear and with a different functional relation between the different ecosystem condition indicators and ecosystem services supply. These conditions will also be spatially heterogeneous, requiring the use of Geographical Information Systems (GIS) to record ecosystem condition in physical units. A first question is if these different physical indicators should be aggregated to produce one (or a few) composite indicators for ecosystem condition. The weighing and aggregation of individual indicators to establish a composite indicator is not straightforward (e.g. Weber, 2007; Stoneham et al., 2012).

   A second issue is what benchmark condition to select for ecosystem condition. In some countries there are defendable and clearly defined conditions that can be selected, such as the land cover and ecosystem condition prevailing in Australia prior to European settlement (e.g. Stoneham et al., 2012). However, in many countries ecosystems have been modified by people since centuries, and a benchmark based on physical criteria is difficult to select. In this case the alternative is to select a specific, arbitrary year, for instance the year in which the ecosystem accounts start.

   A third issue is how to deal with other factors that influence ecosystem condition and ecosystem output, such as natural disasters, natural fluctuations in environmental conditions (such as annual rainfall and rainfall distribution) and changes in ecosystem output due to different ecosystem management. Moreover, ecosystem degradation does not necessarily lead to a reduced output from the ecosystem, which also depends on how the ecosystem is managed, for instance a loss of soil fertility may be compensated by an increased use of inorganic fertilizers.

ii. *Measuring degradation in monetary terms*. There are two main alternatives to assess the costs of degradation (i) valuation on the basis of the cost that it would take to restore the ecosystem to its reference/benchmark condition; or (ii) valuing ecosystem degradation in terms of a reduction in capacity to supply ecosystem services. There are several proposals for applying the restoration cost approach. For instance, the SEEA 1993 proposes to simply subtract restoration costs from GDP in order to arrive at a 'green GDP'. This proposal was criticized by for example Harrison (1993), who argues that the effects of degradation may already be included in the form of reduced output and subtracting them again would therefore amount to double counting. A second alternative (Harrison, 1993; Vanoli, 1995) proposes to add these degradation costs to GDP, as these restoration costs should not be interpreted as measures of degradation, but rather as a representation of the value of environmental services provided free to the economy. At the same time they would be recorded as a degradation of natural capital, with the result that NDP would remain identical. A third alternative (Vanoli, 1995) emphasizes that these degradation costs measure our 'over-consumption' of nature (they present externalities which are not internalized). In accounting terms this would result in negative savings, which are balanced by a capital transfer from the environment to the economy. Recent proposals made by the European Environment Agency for estimating a consumption of ecosystem capital (EEA, 2010; Weber, 2007) resemble the latter approach.

    However, there are a number of principal concerns with the different variations of the restoration cost approach. First of all, they measure *ex ante* hypothetical costs, which is at odds with a transaction based *ex post* accounting system as the SNA. Moreover, actual restoration may lead to price changes which would impact behavior and as a result change economic activities. Second, on a fundamental level what seems to be missing is an argument why this previous state would need to be restored. For example, perhaps society has willfully decided to change a forested area into a city due to population growth. Third, combining restoration costs with the concept of ecosystem services supply is problematic: restoration costs may in some cases be only weakly correlated with gains in ecosystem services supply.

The second option is to analyze ecosystem degradation or rehabilitation on the basis of changes in the capacity of the ecosystem to supply ecosystem services. The advantages are that this approach does not require the establishment of a reference benchmark. Moreover, it allows differentiating between ecosystem services supply *by* ecosystems and degradation *of* ecosystems. The disadvantage of this approach is that we need to make assumptions about the level of service flows that we expect in the future, which will depend on insights in the sustainability of current use patterns and assumptions on how the flow of ecosystem services will change as a function of future ecosystem management. In terms of valuation, degradation may be assessed through the change in net present value of ecosystem services supply during the accounting period, which necessitates assumptions regarding future ecosystem management patters, the applicable discount rate and development of future prices.

iii. *Attributing the costs of degradation to institutional actors.* A third main issue when integrating degradation costs into an accounting system concerns the allocation of degradation to institutional sectors. The sector causing the degradation is often different from the sector affected by the degradation. For example, in many instances of degradation (think of an oil spill) it is likely that the output of the activity causing the pollution is not itself affected but rather other industries' output (for instance fisheries). A related issue concerns whether rehabilitation should be recorded in the production account as output of the relevant economic sector (and hence change GDP) or only in the capital accounts (as in the SNA) as other changes in volume.

Proposals. There is such large variation between ecosystems that ecosystem degradation indicators need to be specified for each specific country and each specific ecosystem type. Aggregation of physical indicators, or the selection of one or a few overarching ecosystem indicators may be possible for some ecosystem in some countries, depending on the ecosystem, data availability and the scientific understanding of the dynamics of the ecosystem involved (Dominati et al., 2010). Sometimes several aspects of degradation can be simultaneously captured with one or a small set of key indicators, as for instance rain-use efficiency in the case of semi-arid rangelands (e.g. Hein et al., 2011). Also the selection of an appropriate benchmark/reference condition will strongly depend on the individual country and perhaps within the country on the ecosystem type involved. In general, it seems most straightforward to set the start of the accounting series as the reference condition, with some countries perhaps having sufficient data availability to select a year in the past, say 2000, as benchmark condition.

Furthermore, a reduction in capacity can be due to either changes in the extent of the ecosystem (which will be reflected in changes in land cover) or changes in the condition of ecosystems (e.g. loss of species). When compiling an ecosystem asset account it will be important to separate these two causes. According to our proposal, and contrary to the reference benchmark approach, conversions of land need not imply degradation.

In terms of valuation, we propose to use changes in the net present value of ecosystem services supply during the accounting period as an indicator for degradation or rehabilitation of the ecosystem. Compared to using restoration costs, this approach is consistent with the accounting principles of the SNA, in the sense that it defines degradation similar to how depletion is defined in the SNA (UN et al., 2009, para 12.26). A complicating factor is that changes in the value of an individual ecosystem service may be due to external factors such as reduced fishing due to a fall in expected output prices, which one usually does not associate with degradation (but on the contrary with a recovery). A necessary condition for monetary degradation should be a decline in relevant physical indicators of condition.

In addition, this recording system can properly account for deliberate changes in ecosystem cover or condition. For instance, conversion of a forest to agricultural land will generally involve a loss in biodiversity and naturalness of an area, and the restoration costs method can be applied, but this conversion may result in more profitable land use, even if all ecosystem services have been considered, and it would be unrealistic to record land use change by default as 'degradation'. Recording degradation in terms of changes in capacity to supply ecosystem services also facilitates recording both negative and positive changes (e.g. as a result of direct investments or indirectly through rehabilitation as a result of lower ecosystem usage) in ecosystem capital.

Furthermore, we propose to use current management as the basis for establishing the future flow of services (even though this may not be the optimal or the sustainable way of managing the ecosystem). The motivation for this choice is that (i) this would be consistent with valuation practices of mineral and energy resources in the SEEA CF (UN et al., 2012), which take current management practices (e.g. resulting in an estimated future extraction path) as point of departure, rather than the optimal value; (ii) this avoids assumptions or complex modeling exercises on the optimal or sustainable management of ecosystems, which may be difficult to assess given that there may exist multiple sustainable states. In practical terms, an assumption that is often made is that (smoothed) prices (or unit resource rent) remain constant (e.g. Veldhuizen et al., 2009). For the discount rate, a social discount rate may be chosen.

In terms of its incorporation in accounts, we propose to record rehabilitation of the ecosystem as a new type of flow 'appreciation of natural capital' in the production account (Vanoli, 2005 p.339, discusses a comparable proposal regarding the recording of discoveries and reassessments of non-renewable resources), rather than as other changes in volume as suggested in the SEEA CF in case of natural growth of biological assets or as output of the relevant economic sector as suggested during the SEEA revision process. The effect of this adjustment would be that GDP would not be adjusted, but that NDP would increase. This would allow for a symmetrical treatment of degradation and rehabilitation. Moreover, a treatment as appreciation avoids associating a value of output with refraining from productive activities.

In terms of allocating degradation to different sectors, we propose to charge the degradation costs in the current accounts on a polluter pays principle, but transfer these same costs in the allocation of income accounts in order to make explicit whose assets bear the impact of the degradation (see also Skånberg, 2001). In the Appendix these recording mechanisms are illustrated for a hypothetical example.

Valuation consistent with accounting principles

Key issues. When discussing valuation principles, it is always important to be clear about the context of use. As we have seen, green accounting is traditionally concerned with measuring (social) welfare whereas environmental accounting strives to provide measures of economic activity. A lot of the work on the valuation of ecosystem services is inspired by the objective to assess the value of ecosystems in terms of generating societal welfare. Such valuation exercises have important uses in informing policy for instance when performing costs benefit analyses or when analyzing alternative land uses. The context of use has important ramifications for the value concept that is required. The SNA is based on transactions and follows in principle market exchange values which exclude consumer surplus (UN et al., 2003). This means that when the objective is to integrate ecosystem service values into national accounts only methods that yield values which are consistent with SNA principles are feasible.

Proposals. There are a number of approaches to pricing ecosystem services that are compatible with the SNA valuation principles. In the case of provisioning services, monetary estimates can often be obtained by looking at the market price of a provisioning service – if traded on a market, or by analyzing the contribution of the ecosystem (i.e. the ecosystem service) to a good that is traded on the market. For instance, in the case of timber, the contribution of the ecosystem is to supply a standing stock of timber that can be harvested. The physical volume of the ecosystem service is the harvested timber quantity. This particular ecosystem service may be valued directly, i.e., by analyzing the price paid for standing wood (i.e. before harvesting) in the local market. It may also be valued by analyzing the market price of harvested timber and deducting harvesting and, if relevant, transportation and processing costs. In an efficient market, these two approaches should yield a comparable price. In some cases, there is no equivalent trade of the ecosystem service itself. For instance, in the case of marine fisheries, there is no market trade of fish stocks prior to harvesting. In this case, the only approach available is to deduct harvesting costs (labor, fuel, depreciation of equipment, etc.) from the value of the landed fish in order to get a net unit price of the ecosystem service (i.e. the fish as caught). This approach is based on the principle of using a unit resource rent for valuing a provisioning service: the unit resource rent represents an estimated price for the ecosystem service. However, a number of market conditions must be in place for estimates of unit resource rent to accurately reflect a price for the ecosystem service. These conditions include that the resource is harvested in a sustainable way and that the owner of the resource seeks to maximize the resource rent (UNSD, 2013). Often, these conditions are not met. In particular, in case of an open access situation the marginal unit resource rent tends to approach zero thus implying that the price of the ecosystem service is zero. Further research is required to develop valuation approaches for ecosystem services harvested under open access conditions, potentially through relating these services to substitute services (for instance fish raised in aquaculture systems).

For regulating services, a valuation method that is potentially aligned with the SNA valuation principles is the replacement cost method. This method is not preference-based and does not provide a measure of the surplus generated by a service (National Research Council, 2005). Instead, the replacement cost method estimates the value of an ecosystem service based on the costs of mitigating actions if the service would be lost. For instance, the costs of a water purification plant can, in principle, indicate the benefits obtained from the water filtration service of an ecosystem (e.g. Hein, 2011). Preconditions for using this method are that the alternative considered provides the same services, is the least-cost alternative, and that it can be reasonably assumed that society would chose to replace the ecosystem service if lost (National Research Council, 2005). A related method is the 'costs of treatment method', which involves estimating the value of an ecosystem service based on the costs of repairing damages that would occur in the absence of the service (National Research Council, 2005). This service is relevant for the erosion and sedimentation control and the air purification service. For instance, in the absence of erosion control, the barrier lake of a hydropower dam would receive higher sediment loads, and the costs of removing these sediments can be used as an indication of the value of the service, under the same conditions of being an adequate and least-cost treatment, and it being likely that society would chose to conduct the treatment if the damage occurs (UNSD, 2013).

The last decade has seen a strong increase in the number of markets for ecosystem services that has been set up. These markets have been developed, in particular, for various regulating services such as carbon sequestration, water regulation and erosion control (Wunder et al., 2008). Most of the markets function at the local or national scale, but for carbon there is a growing global market for carbon sequestration, and for projects leading to reduced emissions of carbon from deforestation or forest degradation (REDD+) (Milos and Kapos, 2008; Peters-Stanley and Hamilton, 2012). Where these markets function efficiently, the price levels provide an indication of the exchange value of the ecosystem services involved (UNSD, 2013). In using price estimates from ecosystem services markets, it needs to be considered, however, that prices in these markets are in part determined by the institutional design and regulatory setting of the market mechanism, and that prices can change rapidly in response to changes in these settings.

A novel, alternative approach with particular relevance for ecosystem accounting is the simulated exchange value approach (Campos and Caparrós, 2011). The approach aims to measure the income that would occur in a hypothetical market where ecosystem services are bought and sold. It involves estimating a demand and a supply curve for the ecosystem service and then making further assumptions on the price that would be charged by a profit-maximizing resource manager. The method analyzes the hypothetical revenue associated with this transaction (but not the associated consumer surplus) in order to estimate the value of the ecosystem service.

A range of other valuation methods for non-market ecosystem services have been developed and applied in the environmental economics literature (see e.g. TEEB, 2010 for an overview). They can be broadly divided into revealed preference and stated preference methods. Although many of these valuation methods include elements of consumer surplus, it may be possible to use demand curves from such studies to estimate an output that is consistent with SNA principles. However, in general, great care needs to be taken when value estimates from the environmental economics literature are used in the context of ecosystem accounting, in particular in a benefit transfer context (e.g. Plummer, 2009), since a range of valuation approaches have been applied and many of the value estimates may not be consistent with the valuation principles of the SNA (UN et al., 2009).

4.4 Discussion

Our proposed conceptual framework has the same point of departure as recent contributions (e.g. Banzhaf and Boyd, 2012 and Bateman et al., 2010) in that ecosystems are broadly speaking conceived as assets (a form of capital) that provide ecosystem services. Our definition of ecosystem services – contributions of ecosystems to productive or consumptive activities – is in line with the definition provided by Banzhaf and Boyd (2012, p.4), who stress two conditions: (1) "ecosystem goods and services are ecological commodities, measured in physical terms"; (2) "ecosystem goods and services are the "final products" of natural systems [...] enjoyed directly by people [...] or productively used in the creation of man-made goods [...]" We are in agreement with the first condition, although we prefer to speak of contributions rather than commodities as the latter has a very material connotation. Regarding the second condition, our conceptual framework also implies that the scope of measurement for ecosystem services includes only *final* contributions. However, contrary to Boyd and Banzhaf who stress the fact that ecosystem services are the 'final products of natural systems', we state that ecosystem services are generally the result of an interaction of ecological processes and human management of an ecosystem, and that in practice the natural component of the specific contribution of the ecosystem can usually not be disentangled with an accuracy adequate for accounting purposes (for instance, separating the specific contribution of ecological processes such as nutrient retention, earthworm activity, water storage, etc., from human management such as fertilizer input, irrigation etc. in the production of crops is usually not practically feasible for large areas of agricultural land (e.g. Barrios, 2007)).

The proposal to extend the production boundary can be found in several accounting proposals (Vanoli, 1995; Peskin, 1976; EEA, 2010, Harrison, 1993). We propose however distinct recording mechanisms for ecosystem services with a public goods character such as many regulating services (where human intervention is not needed to extract or harvest the ecosystem service) and services with a private character such as most of the provisioning services (building on the work of Campos and Caparrós, 2006). Services that are already included in standard output are separately identified in order to recognize their contribution to production.

We consider ecosystems as spatially explicit and mutually exclusive assets. For the purpose of accounting, they can be delineated in terms of land cover units. The functioning of ecosystems is determined by their components and interactions between these components (Levin, 1992), and it is not meaningful to disentangle the impact of the large number of individual ecological processes and ecosystem components (soil, land, vegetation) to the generation of ecosystem services: most if not all ecosystem services depend on the functioning of the ecosystem as a whole (MA, 2003; TEEB, 2010). Our framework recognizes that the large majority of ecosystems have been modified by people (MA, 2003). In accounting terms this implies that due to the extension of the production boundary the distinction between cultivated and natural biological resources of the SNA ceases to exist. Herewith our proposal differs from the asset classification as proposed in SEEA Central Framework (UN et al., 2012) in which the individual components of ecosystems (land, soil, water) are identified as non-produced assets, however, our ideas are in line with the extended asset boundary of SEEA.

We analyze ecosystem degradation and rehabilitation on the basis of changes in the capacity of the ecosystem to supply ecosystem services. In monetary terms degradation can be estimated by estimating changes in the net present value of ecosystem services supply. Although such an approach is data intensive and requires assumptions about future ecosystem management, it is consistent with SNA principles. Also, our proposals allow transferring benefits and degradation costs between sectors. The main rationale for such transfers is the fact that benefits and costs may accrue at places different from where they were initially generated. As a result, a comprehensive picture emerges about who benefits from ecosystem services, who is responsible for eventual degradation, and who bears the costs. Another novelty is to record ecosystem rehabilitation as appreciation of natural capital in the production account, hereby obtaining a symmetrical treatment with degradation.

It is likely that more challenges will surface at the time of the actual implementation of ecosystem accounts in specific countries. One of the main challenges in this regard is data availability. Some 20 to 30 different types of ecosystem services have been distinguished in the literature (e.g. MA, 2003; TEEB, 2010), and each service requires a different dataset. Moreover, these data need to cover both the flows of ecosystem services and the stocks/condition of ecosystems providing these services. Since most of these data have a spatial component, i.e. differ from one location to the next, the data will need to be spatially referenced in a GIS. Data availability is also likely to differ strongly for the various services, it can be expected to be relatively high for many provisioning services and much lower for most regulating and cultural services. Finally, information on the value of non-market ecosystem services may not be comprehensive, which means that some degree of interpolation may be required. Even though there is increasing experience with the meta analysis of the values of ecosystem services (e.g. Brander et al., 2006), there may be insufficient information to reliably assess the value of all non-market ecosystem services. Hence, the application of the concepts outlined in this and other papers will always be constrained by and perhaps needs to be adjusted to the availability of data. Hence, in view of these constraints, it may be advisable to start with pilot studies focusing on specific ecosystem services for which relatively ample data are available, or to test ecosystem accounting first at the sub-national (e.g. provincial) level. Clearly, in case not all services are included, the ecosystem account does not provide a complete picture of ecosystem services or capital.

In addition, there are a number of important issues that we did not analyze in our paper. A first issue is the treatment of biodiversity. One could argue that biodiversity services are internal to ecosystems much akin to supporting services (as in MA, 2003). However, people also value species diversity and/or the protection of rare species independent of the role of these species in supplying ecosystem services (e.g. Mace et al., 2012). This value is generally seen as a non-use value (see e.g. Pearce and Moran, 1997; Pearce, 2007), which is therefore difficult to reconcile with the valuation approach of the SNA. A second issue deals with ecosystem resilience in relation to ecosystem capital. We have defined ecosystem capital as the capacity of ecosystems to generate ecosystem services, now and in the future. A highly resilient ecosystem recovers faster following disturbance and/or is less affected by a certain amount of disturbance – and therefore has a higher capacity to generate ecosystem services, depending on the amount of disturbance (e.g. from climate change) that the ecosystem will experience. However, this higher capacity is only apparent at the time of, or following a disturbance. Even though there may be specific ecological indicators that indicate ecosystem resilience (see e.g. Carpenter et al., 2001; Briske et al., 2010), the disturbances itself may have a probabilistic nature and be hard to predict a priori. Therefore the impact of a loss of resilience on ecosystem capital may be difficult to assess, and also given the remaining uncertainty on ecosystem resilience in most ecosystems (Trush et al., 2009), this aspect may be difficult to include in ecosystem accounts at this point in time. Finally, it needs to be kept in mind that ecosystem accounts are no panacea. First, it will take a number of years before they will be operational – which is conditional on adequate resources being made available for ecosystem accounting. Second, data constraints mean that it may prove elusive, at least in the coming decade, to capture all ecosystem services in such accounts, in either biophysical and/or monetary indicators. Third, there are a range of environmental issues that are not adequately dealt with in ecosystem accounts. For instance, national level ecosystem accounts are not suitable to effectively record water flows in international watersheds, or coarse scale climate change effects such as changes in ocean-atmosphere interactions. Hence, both the potential and the limitations of ecosystem accounting need to be kept in mind, both while developing the accounts themselves and when engaging in the wider debate on ecosystem accounting with policy makers and the public.

4.5 Conclusions

There is an increasing interest in the integration of ecosystem capital and ecosystem services into national accounting systems such as the SNA or SEEA. Due to the fact that many ecosystem services are non-market and that degradation costs are often not internalized, current accounting practices provide an incomplete picture. Many of the benefits supplied by ecosystems are already captured, implicitly, in the SNA, for instance most provisioning services and those regulating services supporting economic activities that are measured in the SNA. The additional information revealed by ecosystem accounting pertains to the dependency of economic activities on ecosystems, the status of ecological capital, and the rate and effects of ecosystem degradation (or rehabilitation). Given the ongoing degradation of ecosystems world-wide (e.g. MA, 2005; TEEB, 2010), it is important that ecosystem accounts are urgently developed in order to provide a measurement framework for informed decision making. This paper contributes to this development and the broader debate on ecosystem accounting by analyzing potential solutions for four key challenges in ecosystem accounting, respectively defining ecosystem services in an accounting context, allocating ecosystem services to institutional sectors, recording ecosystem degradation, and monetary valuation. In spite of recent progress in the field of ecosystem services modeling (e.g. TEEB, 2010) and ecosystem accounting (e.g. UNSD, 2013), further steps are needed before ecosystem accounting can be operationalized in a comprehensive manner (i.e. including a representative and comprehensive array of ecosystem services). This will require the further development of specific SNA-consistent recording methods for ecosystem services and ecosystem capital, and the scaling up of efforts aimed at monitoring ecosystem services use at aggregated (e.g. provincial or national) scales.

4.6 Appendix

Recording ecosystem services and degradation in the sequence of accounts

We revisit the hypothetical economy that was introduced in Section 4.3 in order to illustrate the recording of ecosystem services in more detail, throughout the sequence of accounts, as shown in Table 4.2.[2] First of all, an additional line non SNA output is introduced in the production account, which covers output of ecosystem services that are not reflected in standard output such as regulating services and amenity services. In the case of carbon sequestration due to its public goods character we propose to record this in an additional sector ecosystems, while the provisioning services continue to be recorded in the traditional sectors (in the example here, the land owner). As a result we have as in Table 4.1 a value added of 190 for our economy, however, compared to Table 4.1, the allocation is different: 160 accrues to the land owner and 30 to the ecosystem sector.

The generation of income account is modified in order to explicitly identify services that are already included in standard measures of economic output in terms of their contribution to the gross operating surplus (provisioning services in the example here).[3] As a result it becomes visible that the land owner benefits from non-marketed provisioning services.

In reality, ecosystems do not own bank accounts and cannot consume. Therefore, the imputed (fictitious) income that they earn by producing ecosystem services is transferred to other sectors (in our example households) in the distribution of income account, by introducing a new type of flow entitled 'ecosystem transfers'. Households subsequently consume these services in the use of income account (the example shows the sector households as beneficiaries of carbon sequestration) paid for by the additional income they receive from the ecosystem sector. The overall effect of these proposals is that the final consumption of households is increased, but their savings remain unaltered, compared to standard SNA recordings.

Degradation costs are introduced in the production accounts diminishing the value added of land owner assumed to be responsible for the degradation. These costs are subsequently transferred between the polluter and the pollutee in case these are units in different sectors. This has the effect that in the balance sheet the degradation costs accrue to the sector who is impacted by the actual damages, reducing its production potential in the next period (in the example this is the ecosystem sector).

[2] In fact Table 4.2 only illustrates recordings in a subset of the full sequence of accounts (see UN et al., 2009, Annex 2 for an overview of the complete sequence of accounts).
[3] A similar breakdown – but in the production account - was proposed during the 2008 SNA revision process (Ahmad, 2004) for capital services.

We can check that the system is balanced as total supply (output of 230) equals total use (40 intermediate consumption plus 190 final consumption). It is important to realize that the opening and closing stocks of the balance sheet of the hypothetical economy are affected due to the extension of the production and asset boundary. The overall macro effect of our proposals is that compared to standard accounting conventions, the non-market output from ecosystem services is added to the total output and hence value added of an economy, while the costs of ecosystem degradation are subtracted

Table 4.2 Recording of ecosystem services in sequence of accounts (fictional data)

	Land owner	Consumers	Ecosystems	Total
Production and generation of income accounts				
Output - SNA	200			200
Output - non-SNA			30	30
Intermediate consumption - SNA	40			40
Intermediate consumption - non-SNA				
Gross value added	160		30	190
Compensation of employees	50			50
Gross operating surplus	110		30	140
provisioning of crops	*80*			80
capital services	*30*			30
Consumption of fixed capital	10			10
Degradation	15			15
Net operating surplus (adjusted)	85		30	115
Distribution of income account				
Compensation of employees		50		50
Ecosystem transfers		30	–30	0
Degradation transfers	15		–15	0
Net disposable income (adjusted)	100	80	–15	165
Use of income account				
Final consumption - SNA		160		160
Final consumption - non-SNA		30		30
Net saving (adjusted)	100	–110	–15	–25
Balance sheet (non-financial assets)				
Opening stock	1,000		300	1,300
consumption of fixed capital	10			10
Degradation			15	15
Closing stock	990		285	1,275

5.
A review of World Bank estimates of wealth for the Netherlands

5.1 Introduction[1)]

The change in a country's national wealth over time provides an indication to what extent its development is being sustainable (World Bank, 2011; UNU-IHDP and UNEP, 2012). National wealth consists of the sum total of different types of capital (produced, natural, financial, human, etc.) available to a country (World Bank, 2006). Broadly speaking, development is considered sustainable as long as a country's capital base is being maintained across generations, although opinions differ on the substitutability between different forms of capital (weak vs. strong sustainability; Neumayer, 2003). Another issue concerns the treatment of population growth and whether capital preservation should be assessed on a per capita basis (Arrow et al., 2003b). While the initial focus in this area was on assessing changes in wealth through (net) investments, which resulted in well-known indicators of sustainability such as genuine savings (or adjusted net savings, ANS); Hamilton 1996) or policy prescriptions for achieving sustainability such as Hartwick's rule (Hartwick, 1977), in recent contributions (World Bank 2006, 2011; UNU-IHDP and UNEP, 2012) the objectives have broadened towards obtaining estimates of the various capital assets that are the constituents of wealth, an area which we refer to as wealth accounting.

Empirically, one may differentiate between two approaches towards compiling wealth accounts. The residual approach – called a 'top-down' approach by Hamilton and Liu (2013) – associated with the World Bank (2006; 2011) starts by estimating total wealth (usually called 'comprehensive wealth') based upon the present value of sustainable future consumption. Subsequently, intangible or human capital is calculated as a residual by subtracting from total wealth capital values such as produced capital that are known directly. An important element in the approach is the assessment of sustainability of consumption using the ANS indicator. The approach is grounded in an underlying macro-economic model, as described in Hamilton and Clemens (1999) and Hamilton and Hartwick (2005). The second approach (Arrow et al., 2012; Dasgupta, 2009; UNU-IHDP and UNEP, 2012) – called a 'bottom up' approach by Hamilton and Liu (2013) – attempts to derive wealth estimates by summing up the shadow values for all individual assets

[1)] The author would like to acknowledge Cees Withagen for valuable comments, as well as several colleagues from Statistics Netherlands, in particular Mark de Haan, Rutger Hoekstra and Erik Veldhuizen for commenting on an earlier draft. In addition, research done for this chapter has resulted in a letter to the editor of JEMA (Edens, 2013b) regarding the article of Ferreira and Moro (2011), which has been summarized in Section 5.5. The author would like to thank Susana Ferreira and Mirko Moro for their willingness to discuss their article, and Mike Holland for discussing the issue of SO_2 damages. I would also like to acknowledge Kirk Hamilton for his valuable suggestions for improvement.

of a country including human capital (sometimes called 'inclusive wealth').[2] As pointed out by Hamilton (2012) who made a comparison of the wealth estimates of both approaches for the United States, there is a large discrepancy in results due to a different choice of the asset boundary: 'health capital' is included in inclusive wealth, but not in comprehensive wealth.

In between these two approaches lie efforts by statistical offices that add direct estimates of human capital to asset values available from the national accounts' balance sheets in order to estimate total wealth (Gu and Wong, 2010; Statistics Netherlands, 2012). A variation of the residual approach is followed by Graeker (2008) and Alfsen and Greaker (2006), who use net national income to estimate a return to human capital after subtracting resource rents and other incomes. A key issue in both approaches is how to reconcile official data on asset values with the need to consistently estimate data for a large panel of countries, relying on generic assumptions.

Recently, the World Bank (2011) presented time series for its comprehensive wealth estimates in constant prices for over 120 countries, which is a major step forward compared to the single year estimate in World Bank (2006). In addition, several methodological improvements were implemented such as the inclusion of separate estimates for net foreign assets. According to the World Bank's findings "*Intangible wealth is the largest single component of wealth in all income groups and the fastest growing one as well… Most countries start out with relatively high dependence on natural capital. They use these assets to build more wealth, especially produced capital and intangible (human and institutional capital).*" The World Bank argues that development can be conceived as "*a process of building and managing a portfolio of assets*" (World Bank 2011, p.4). It is therefore important for policy making that robust estimates of the various components of wealth are obtained.

The purpose of this chapter is to review the wealth and ANS estimates of the World Bank for the Netherlands. Unfortunately, UNU-IHDP and UNEP (2012) does not contain inclusive wealth estimates for the Netherlands, so a comparison could not be made. Obviously, as the World Bank attempts to compare wealth across such a large number of countries, restrictions of data availability and reliability of data for some countries, places a limit on detail that can be used. The paper therefore also attempts to refine these estimates using additional data sources whenever

[2] Dasgupta (2009) discusses possible names for this conception of wealth as the sum of assets valued at shadow prices and states that he is "hoping that the term "comprehensive wealth" will prevail, because it is vivid". However, in Chapter 1 of UNU-IHDP and UNEP (2012; written by Dasgupta and Duraiappah) this conception of wealth is called 'inclusive wealth' in order to distinguish it from the World Bank's approach, which is referred to as 'comprehensive wealth'.

available for the Netherlands.[3] Given the impact the World Bank's results have on both policy and academia, it is important to get an understanding of their robustness.

The Netherlands is a good candidate for such a review, as Statistics Netherlands compiles already comprehensive balance sheets in both current and constant prices for produced and non-produced assets such as land and energy resources as part of its national accounts. The Netherlands has an extensive environmental accounting program which covers air emissions accounts and environmental protection expenditure (Statistics Netherlands, 2010b), which are both required data sources for the World Bank's approach for estimating comprehensive wealth. In addition, the Netherlands has compiled as part of its productivity measurement program estimates on various types of intellectual property products, which are a subset of what the World Bank refers to as intangible capital. Furthermore, in 2012, experimental estimates for human capital have been compiled, which allow comparing wealth estimates obtained by the 'residual' approach with direct estimates (a similar approach is followed Hamilton and Liu, 2013).

The outline of the chapter is as follows. In Section 5.2 the approach to wealth accounting of the World Bank (2006; 2011) will be discussed and its relationship to ANS clarified. Section 5.3 presents our results. We compile a time series of ANS estimates for the Netherlands and compare its components with World Bank estimates. We present a time series of wealth estimates in constant prices based upon the residual approach. We then discuss differences in methods and data sources with the World Bank for the year 2005 and compare residual estimates with recent direct estimates of intangible capital. We also provide a sensitivity analysis for the comprehensive wealth estimates. Section 5.4 discusses the key issues inherent in the comprehensive wealth approach and provides suggestions for future improvements. Section 5.5 contains a discussion of Ferreira and Moro (2011), who have done a similar exercise for genuine savings in the case of Ireland. Unfortunately, as is shown their article has some shortcomings. Section 5.6 concludes.

[3] This is in line with the position of the World Bank: "Although the World Bank will continue to compile and improve global wealth accounts, the eventual goal is for countries to implement wealth accounting themselves under standard guidelines. Compared to intergovernmental organizations, countries have much greater resources and access to information that enables them to compile more accurate and comprehensive wealth accounts." (World Bank, 2011).

5.2 World Bank's approach to wealth accounting

The World Bank's approach to wealth accounting (World Bank 2006, 2011; Hamilton and Hartwick 2005) consists of estimating the following identity, which holds for a competitive economy whose production function exhibits constant returns to scale (Hamilton and Liu, 2013):

$$W_t = K_t + H_t + N_t + F_t = \int_t^{\infty} C(s) e^{-r(s-t)} ds \qquad (1)$$

with

W_t	comprehensive wealth;
K_t	produced capital;
H_t	intangible capital;
N_t	natural capital;
F_t	financial capital (net foreign assets);
$C(s)$	consumption in year s;
r	the social discount rate (assumed constant).

That is, comprehensive wealth W can be measured either by adding up asset values for produced (K), natural (N), financial (F) and intangible capital (H), or alternatively by measuring the present value of future consumption. In general values of K, N and F in current prices are known from the balance sheets of the national accounts, but H is not. In the System of National Accounts (UN et al., 2009) human capital – one of the main components of intangible capital – is not considered an asset, and therefore not valued. In the World Bank's approach H is therefore not measured directly but is derived as a residual by subtracting K, N and F from comprehensive wealth that is estimated by measuring the present value of consumption using the right hand side of Eq.1. As a result, the World Bank's residual category H – intangible capital – is a mixture of various types of assets: along with human capital also social capital or institutional capital and natural capital assets that are not explicitly included under N due to data shortages (e.g. the value of a country's fish stocks) are included.[4)]

[4)] World Bank usage of the word 'intangible capital' is at odds with its usage in the 1993 SNA. The World Bank uses the term intangible capital to refer to all non-physical, non-financial assets, hereby excluding so-called intangible fixed assets (e.g. mineral exploration, software). At the same time, the World Bank's category is much broader as it also includes forms of capital such as human and institutional capital that are not considered assets in the SNA. However, the latest version of the SNA – the 2008 SNA (UN et al., 2009) – no longer uses the distinction tangible/intangible, and introduces the concept of Intellectual Property Products to cover intangible fixed assets.

Empirically, in order to estimate wealth as the present value of sustainable consumption, we need to know the future consumption path. Rather than constructing a comprehensive model of the economy and calculating the optimal consumption path, the World Bank assumes that the present level of a country's consumption is on a sustainable path. Here ANS acts as a boundary condition to assess whether current consumption is in fact sustainable (World Bank, 2006, p.144). When negative savings occur, they are subtracted from consumption as available in the national accounts. In order to estimate the right hand side of Eq. 1, we therefore first need to estimate ANS to assess whether they are positive.

Important when evaluating Eq. 1 is the choice of a discount rate. The World Bank uses the Ramsey formula for the discount rate (Eq. 2), assumes that η =1 and that the growth rate of consumption is constant.

$$r \equiv \rho + \eta \dot{C} / C \tag{2}$$

with
ρ pure rate of time preference;
η elasticity of utility with respect to consumption.

As a result, future consumption is discounted at the pure rate of time preference ρ, taken to be 1.5%.

The model that underlies the definition of adjusted net savings is presented in Hamilton and Clemens (1999) and is described here in the Appendix. It results in the well-known expression of ANS as gross savings (GS) minus the depreciation of produced capital (D_K) minus the depletion of natural resources (D_N) and degradation costs (D_E) plus investments in human capital (A_H):

$$ANS \equiv GS - D_K - D_N - D_E + A_H \tag{3}$$

The value of gross savings of an economy is equal to its gross disposable income minus consumption. It takes also (net) income earned by residents abroad into account, for instance interest payments on foreign owned assets.

However, use of the model also implies that C in Eq. 1 is not defined as the standard concept of consumption in the System of National Accounts C_{SNA} as education expenditures need to be subtracted, which we will denote as C_{MEW} (this is demonstrated more formally in the Appendix, the subscript stands for Macro Economic Welfare). Current educational expenditure – which as argued by Hamilton (1996) can be seen as a lower-bound estimate of investment in human capital – needs to be moved from the final demand category consumption towards investment. The second issue concerns the treatment of so-called defensive expenditures i.e. expenditures for environmental protection or rehabilitation (see also Section 4.3). The model presented in the Appendix (and as argued by Hamilton 1996) implies that from a welfare perspective such expenditures be considered as intermediate consumption rather than as final demand and therefore would need to be subtracted from consumption. At the same time the value of environmental services would need to be added to consumption.[5)] In the absence of adequate estimates of the value of these services we have decided not to subtract defensive expenditures to avoid a bias in the outcomes, although the effect of subtracting defensive expenditures will be discussed as part of the sensitivity analysis.

Therefore, consistency with the model implies that in empirical applications of Eq.1 C_{MEW} should be used as long as $ANS > 0$. Finally, by differentiating Eq. 1, we also obtain an important result that net income equals the return to wealth, or:

$$C + \dot{W} = rW = C + ANS \tag{4}$$

where the second identity holds when the interest rate is assumed constant. An additional complication to consider in empirical testing of Eq. 4 is that the value of some environmental services (e.g. an amenity service) may already be reflected in current asset prices, while excluded from income measures.

5.3 Results

Adjusted net savings

Table 5.1 provides an overview of the components that make up ANS and allows for an in depth comparison between World Bank estimates and our estimates. World Bank ANS estimates are available at www.worldbank.org (accessed May 2011) which have been converted to Euros.

5) The model in the Appendix assumes that these environmental services impact utility directly, which would compare, as discussed in Chapter 4 where the production boundary is extended, with a situation of ecosystem services as contributions to consumptive activities.

First of all, gross saving and consumption of fixed capital data were obtained from the national accounts (Statistics Netherlands, 2011a). The small difference with World Bank data may be caused by the currency conversions or the use of different vintages of Dutch national accounts data, which are being revised regularly.

Regarding the educational expenses, choices have to be made regarding the scope of expenditures that are taken into account. For instance, whether overhead costs made by the Education Department are included. We have used the 'OECD indicator of educational expenses' published by Statistics Netherlands (2011a). The World Bank uses data from UNESCO. We see that educational expenditures slightly diverge between these two sources especially for recent years.

Larger differences exist between the depletion estimates, which is the combined result of different methodologies as investigated in detail in Chapter 3 (Edens, 2013a) and different data sources. The World Bank (2011) calculates depletion as the ratio of the present value of rents to exhaustion time of the resource, whereas our estimates use the methodology recommended by the SEEA Central Framework (UN et al., 2012) which consists of current extraction volume multiplied by the price of a unit in the ground (averaged between beginning and end of the accounting period). Differences between both methods arise due to different assumptions used in valuing the asset (we assume a linearly decreasing extraction path in line with Dutch Government policy, whereas the World Bank effectively assumes a constant extraction rate to exhaustion) as well as the fact that the constant extraction rate may be different from the actual extraction during the accounting period in question. Moreover, we use a 3 year moving average to estimate the future unit resource rent and furthermore assume that it remains constant. The discount rate used in both calculations is however the same: 4%. In terms of data sources, World Bank uses international data sources, whereas our estimate uses national accounts data on the oil and gas extraction industry (Veldhuizen et al., 2009).[6)]

The World Bank CO_2 damage estimates are on average only 1% higher than our own estimates, although the patterns diverge slightly between years. Differences occur because we use air emissions figures from the Dutch environmental accounts (Statistics Netherlands, 2011) which are on average 5% higher than World Development Indicators data used by the World Bank. We have used for CO_2 damages a value of 20$/tC for the year 1995 to allow comparison to World Bank.[7)]

6) The depletion estimate presented here is restricted to natural gas reserves.
7) Tol (2008) based on a meta-analysis of social cost of carbon reports that 'the median of the Fisher-Tippett kernel density for peer-reviewed estimates with a 3% pure rate of time preference and without equity weights, is $20/tC'.

In Table 5.1 we have also included our estimates for other greenhouse gases such as methane and nitrous oxide.

Table 5.1 Comparison of ANS estimates for the Netherlands by components: World Bank estimates vs. author's estimates (1995-2005)

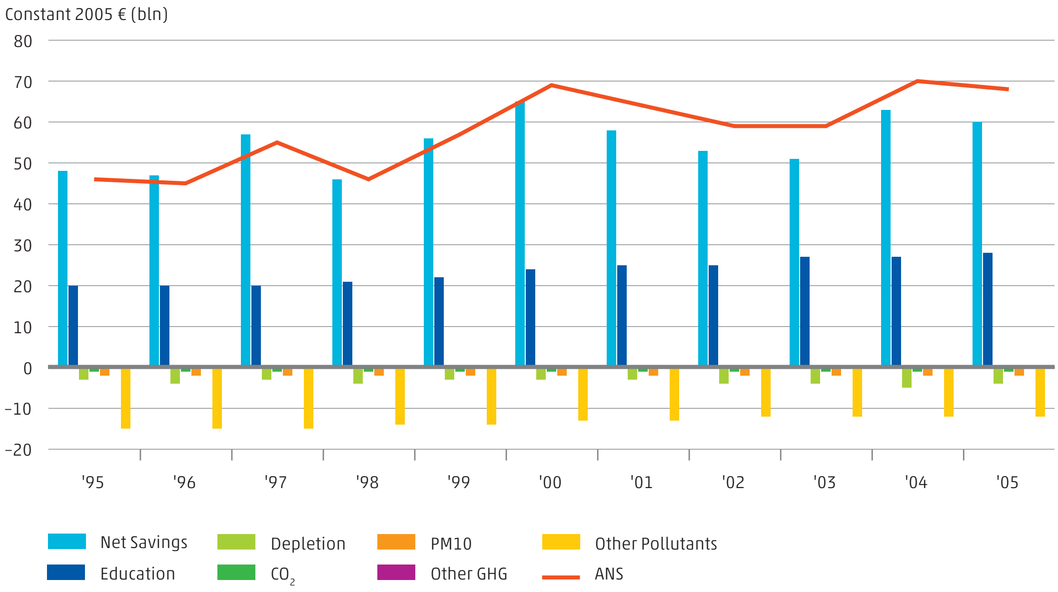
	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
	current billion €										
World Bank											
Gross Savings	81	84	95	89	103	117	118	119	121	134	135
Consumption of Fixed Capital	45	47	49	52	56	61	66	69	71	73	76
Net savings	36	37	45	37	46	56	52	50	50	61	59
Educational expenditure	16	16	17	17	17	19	21	21	23	24	25
Depletion	−2	−3	−3	−2	−2	−5	−6	−4	−5	−6	−9
CO_2 damage	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1
PM_{10} damage	−1	−1	−1	−1	−2	−2	−2	−2	−1	−1	−1
Adjusted Net Savings	48	47	57	49	59	66	64	64	65	76	73
Our estimates											
Gross Savings	83	85	96	91	105	119	120	120	121	135	136
Consumption of Fixed Capital	45	47	49	52	56	61	66	69	71	73	76
Net savings	38	38	47	39	48	57	54	50	50	62	60
Educational expenditure	16	16	16	18	19	21	23	24	26	27	28
Depletion	−2	−3	−3	−3	−3	−2	−3	−4	−4	−4	−4
CO_2 damage	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1	−1
PM_{10} damage	−2	−2	−2	−2	−2	−2	−2	−2	−2	−2	−2
Other GHG	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*	*−0*
Other Pollutants	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*	*−12*
Adjusted Net Savings	36	36	45	39	49	61	59	56	57	69	68
	%										
% GNI	12	11	13	11	13	14	13	12	12	14	13
Ratio WB/our estimates											
Gross Savings	98	98	99	98	98	99	99	99	100	99	99
Consumption of Fixed Capital	100	100	100	100	100	100	100	100	100	100	100
Net savings	96	96	98	94	96	98	97	98	99	98	98
Educational expenditure	100	97	103	95	91	91	91	88	88	88	88
Depletion	84	117	108	74	67	228	196	111	142	146	208
CO_2 damage	88	93	106	104	101	118	119	113	94	87	87
PM_{10} damage	70	74	85	86	89	101	112	112	62	61	62
Adjusted Net Savings	132	130	125	126	120	109	109	113	114	109	107

Data sources: World Bank ANS estimates are available at www.worldbank.org (accessed May 2011) which have been converted to Euros; Statistics Netherlands (2011a) and own estimates.

The largest difference arises by our inclusion of degradation costs for pollutants not covered by the World Bank, such as: NO_x, SO_2, NH_3, PM_{10} and NMVOS. Damage estimates per kg of pollutant were taken from CE (2010) the most recent study available with damage estimates that apply specifically to the Netherlands. These damage estimates have been adjusted for inflation using consumer price index (CPI) weights. The damages caused by these other pollutants are in case of the Netherlands a lot higher than damages for PM_{10} and CO_2: they represent 82% of total environmental degradation costs.

Figure 5.2 shows the time series of adjusted net savings estimates for the Netherlands expressed in constant year 2005 prices using GDP deflators. We see that ANS estimates are positive for all years and more or less increasing, although there are large fluctuations from year to year. Damage estimates are decreasing, which is primarily due to the decrease in non-CO_2 emissions that have been significantly reduced over the past couple of years. Our results corroborate the findings of the World Bank that found positive ANS values for the Netherlands. However, the level of our ANS estimates expressed as percentage of GNI over the period 1995–2005 is slightly lower: 13% compared to 15% according to World Bank (2011), which is due to our inclusion of a larger range of pollutants. Without these additional pollutants our estimate would be 16%.

Figure 5.2 ANS and its components for the Netherlands (1995-2005)



Wealth estimates

Wealth based upon the residual approach

As we found positive ANS estimates for all years, according to the model discussed in Section 5.2, we are able to estimate comprehensive wealth directly as the discounted present value of future consumption in the right hand side of Eq. 1. For purposes of comparison, we use the same assumptions as the World Bank (2011): a discount rate of 1.5%; a time horizon of 25 years, a smoothed 5 year centered average of C_{MEW} data. C_{MEW} data were first calculated in current prices using C_{SNA} data from the national accounts together with data on educational expenditures (Statistics Netherlands, 2011a). The time series of C_{MEW} is subsequently expressed in year 2005€ prices using GDP deflators.

Statistics Netherlands compiles asset values for natural and produced capital in constant prices (year 2000), which have been adjusted to year 2005 prices to ease comparison with World Bank results. Financial assets in principle cannot be decomposed into a price and volume component. Here, the method of the Australian Bureau of Statistics (2001) was followed in which financial assets are deflated by a general consumer price index to correct for the change in purchasing power of the value that these financial assets represent.

Table 5.3 Comprehensive wealth estimates for the Netherlands using World Bank's approach (1996-2005)

	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
	billion 2005 €									
Produced capital	1,383	1,419	1,455	1,495	1,532	1,564	1,589	1,612	1,630	1,651
Urban land	892	896	913	926	936	943	953	963	970	977
Natural capital	196	194	191	187	179	176	174	172	171	169
Financial capital	10	−26	−70	−80	−71	−34	−4	74	158	141
Residual	3,310	3,503	3,749	4,000	4,148	4,245	4,292	4,224	4,145	4,186
Total wealth	5,791	5,986	6,239	6,527	6,724	6,895	7,003	7,044	7,075	7,125
	%									
Relative share										
Produced capital	24	24	23	23	23	23	23	23	23	23
Urban land	15	15	15	14	14	14	14	14	14	14
Natural capital	3	3	3	3	3	3	2	2	2	2
Financial capital	0	0	0	-1	-1	-1	0	0	1	2
Residual	57	58	59	61	62	62	62	61	60	59

Source: Author's calculation based on Dutch national accounts data (Statistics Netherlands, 2011a).

Table 5.3 shows that comprehensive wealth increases, in keeping with the rise in consumption that the Dutch economy experienced. However, it is striking that we find that the relative importance of intangible capital does not increase over time. After a gradual increase, in 2001, its reaches a peak at 62% after which it slightly declines towards 59% in 2005. The time series of wealth in constant prices that we construct does not corroborate the World Bank's narrative that intangible capital increases over time (let alone the 'fastest growing one'). It is remarkable that we find a much lower share of intangible capital than found by the World Bank (around 80%) for the Netherlands.

Comparing wealth components using direct estimates of intangible capital

To investigate the latter issue in greater detail, Table 5.4 compares wealth estimates for different types of capital for the Netherlands in the year 2005 using direct estimates of intangible capital with World Bank estimates.[8] The World Bank provides country data on wealth estimates on their website expressed in U.S. dollars, which have been converted in Euros.[9] Estimates for produced, natural and financial capital are obtained from the balance sheets of the national accounts (Statistics Netherlands, 2011a). For ease of comparison, we have separately identified the value of urban land, which the World Bank registers under produced assets, whereas urban land according to the SNA is considered a non-produced asset. Agricultural land is included under natural capital.

Produced capital estimates by the World Bank are 30% lower than Statistics Netherlands' estimates. The reason being that the World Bank estimates for fixed capital are based upon generic assumptions (World Bank, 2011, p.143) such as a universal depreciation rate of 5% as well as service lives of 20 years, while Statistics Netherlands uses a PIM (perpetual inventory method) model with differentiated service lives and depreciation patterns for different types of assets.

[8] The World Bank does not separately estimate asset values for inventories or consumer durables. Consumer durables such as household appliances, clothes etc. are not included in the balance sheet (only as memorandum item) as the services that they help to provide do not fall within the SNA production boundary. Inventories are assets according to the SNA and therefore included in the balance sheets (just like valuables), however their role in the World Bank's approach is not clear. Therefore, in Table 5.4, inventories and consumer durables are not included.

[9] Using we obtain a value of 7,802 billion euro, which is only slightly higher than the World Bank estimate of 7,786 (0.2% higher), and shows that we have applied World Bank conventions correctly. The small difference in the estimate of total wealth could be due to a number of reasons: national accounts final consumption figures may have changed slightly from the figures that were originally used by the World Bank (2011) study due to updates; rounding errors in exchange rates; or small differences between official Dutch statistics and data in international databases used by the World Bank.

Table 5.4 Comparison of World Bank wealth estimates for the Netherlands with direct estimates (2005)

	World Bank		Direct estimates	
	values	composition	value	composition
	billion €	**%**	**billion €**	**%**
Produced capital	1,160	15	1,651	19
Urban land	278	4	977	11
Financial capital	–22	0	141	2
Natural capital	173	2	169	2
oil and natural gas	93	1	103	1
mineral	0	0	6	0
non-urban land	80	1	60	1
Intangible capital	6,196	80	5,774	66
Human capital			5,493	63
Intellectual Propery Products			281	3
Total wealth	7,786	100	8,713	100

Source: Author’s calculations.

A very large difference is seen regarding the value of urban land, which is less than 30% of the value estimated by Statistics Netherlands. The World Bank estimates the value of urban land of all countries as a fixed percentage (24%) of the value of its produced capital (World Bank, 2006; 2011) using as a benchmark an estimate from Canada.[10)] The value of urban land is estimated by Statistics Netherlands as the difference between the value of fixed assets (i.e. dwellings) as calculated from the PIM model and the statistical WOZ[11)] values (Statistics Netherlands, 2010c) which include the value of the land on which the asset lies. The much higher values of urban land that Statistics Netherlands finds may not come as a surprise given the high population density of the country.

The value for net foreign assets (financial capital) is highly different, which can be explained by the use of different data sources derived from different frameworks. The World Bank uses as data source an updated and extended version of the dataset constructed by Lane and Milesi-Ferretti (2007), whose estimates are close (although they differ about 70%) to official Balance of Payments data (estimated by the Dutch Central Bank). Our estimate is based upon the financial balances of the

10) In fact, World Bank (2011) refers to Kunte et al. (1998), who refer to a report from Statistics Canada (1985), entitled ‘National Balance Sheet Accounts 1961-84’. It is unclear what year (or years) the 0.24 estimate refers to.

11) “Since 1997, Statistics Netherlands has collected data on the average house value in the Netherlands based on the Act on Property Assessment (WOZ). This act was introduced on 1 January 1995 and obliges municipalities to assess the value of all property (WOZ objects), within the municipal borders on a regular basis. The WOZ value is used to impose tax on property owners.” (http://www.cbs.nl/en-GB/menu/methoden/toelichtingen/alfabet/w/woz-value-2.htm)

Dutch national accounts. In theory estimates for financial assets (from the national accounts) and estimates for net foreign assets – sometimes called external wealth – (from the international accounts, or Balance of Payments) should be identical. Both consist of the difference between total assets and liabilities of a country with respect to the rest of the world.[12)] In practice, in case of the Netherlands, there is a large discrepancy between the stock estimates of both data sources. CPB (2009) investigated this discrepancy. Although there are small definitional differences (e.g. different treatment of special purpose vehicles and derivatives) no satisfactory explanation was found, other than that the national accounts use Balance of Payments data as a data source which apparently is greatly changed in the integration process. Other research, that is currently being conducted, suggests that the difference is most likely due to different approaches towards the valuation of foreign direct investments. The reason we prefer to use the national accounts estimate is for consistency purposes: all asset values are derived from the same data source.

Total natural capital estimates are close, although a closer inspection reveals various differences. Subsoil assets are valued at 103 billion € by Statistics Netherlands versus only 93 billion € by the World Bank. Although both methods are based upon the net present value, different estimates are due to various factors as already explained earlier in this Section, most importantly: whereas the World Bank calculates an average unit resource rent for all countries based upon the world price minus production costs, Statistics Netherlands uses direct source information on revenues and costs of the oil and gas industry. Second, the time horizon used by the World Bank is fixed at maximum 25 years for natural gas, whereas Statistics Netherlands uses a specific extraction path consistent with Government policy. Statistics Netherlands estimates values for other minerals resources such as sand and gravel, that are not estimated by the World Bank, at 6 billion €. The value of agricultural land is estimated at 60 billion € by Statistics Netherlands, while the World Bank finds a value of 80 billion € for the combined assets crop land; pasture land; protected areas; timber and NTFR (non-timber forest resources). The latter two values are estimated by the World Bank and not by Statistics Netherlands but are negligible. Protected areas (14 billion € according to the World Bank) are not valued by Statistics Netherlands.

[12)] The breakdown of these assets is however different in both frameworks, for instance Foreign Direct Investment assets would be classified as financial assets (e.g. loans or stocks) in the financial accounts.

Residual versus direct estimates of intangible capital

Statistics Netherlands has undertaken research in the past years that allow comparing the residual estimate of the intangible class of assets with direct estimates of its main components. The difficulty is that intangible capital, as we saw in Section 5.2, is a residual category that consists of different types of capital: human capital, 'missing' natural capital, and social/institutional capital. However, human capital is found to be the largest component of intangible capital (Ferreira and Hamilton, 2010). Statistics Netherlands recently compiled experimental asset accounts for human capital (Statistics Netherlands, 2012, Chapter 8) following the Jorgenson–Fraumeni approach (Jorgenson and Fraumeni, 1989). This approach estimates human capital as the discounted value of expected lifetime labor income. For the year 2006, the value of human capital was estimated at 10.7 times GDP, which – applying the same ratio – would result in an estimate of approximately 5,500 billion € for 2005.

There have been also advances in estimating values for other intangible assets. Statistics Netherlands has compiled preliminary estimates for a broad class of intangible assets (termed "knowledge module"), which includes R&D, firm specific capital, organizational structure, and marketing assets. According to Statistics Netherlands (2009a, p.32), these intangible assets – which are in the 2008 SNA classified as intellectual property products – constitute approximately 17% of total fixed assets (for the year 2006). When we apply this ratio to 2005 data, this would result in approximately 281 billion €. Taken together, the direct estimate of 'intangible capital' would be about 5,774 billion € compared to 4,186 billion € when using a residual approach (Table 5.3). This would imply that following a direct approach, intangible capital would constitute about 66% of total wealth.

When we compare total wealth estimates, we see that the World Bank estimate at 7,786 billion € is a lot higher (about 10%) than our estimate based upon the residual approach which as shown in Table 5.3 amounts to 7,125 billion €. This is because the World Bank resorts to using C_{SNA} rather than using C_{MEW} presumably as this figure is more likely to be available for a large set of countries from international databases. The effect this has on the residual is a downward adjustment of 3% (from 62% to 59%). Our direct estimate of total wealth stands at 8,713 billion € which is about 12% higher than the World Bank's estimate. Interestingly, this would imply that what Hamilton and Liu (2013) call 'the residual of the residual' – i.e. the part of intangible capital that is not formed by human capital – would be negative, rather than the 23% estimated by them. This is due to two reasons: first because the combined value of natural, produced and financial capital is almost twice as high as the World Bank estimate (2,939 vs. 1,590 billion €). Secondly, because the direct estimate of human capital used here is about

20% higher than the result by OECD (2011c) (respectively 10.7 times GDP in 2006 versus 8.3). According to the OECD study the Netherlands would have the lowest human capital/GDP ratio of the 15 countries they considered. The 10.7 value that was found by Statistics Netherlands (2012a) and which is used here is close to the average of all 15 countries reviewed in the OECD study which was a ratio of 10.6. In general one would expect the Statistics Netherlands estimate to be more reliable as it is based on much richer data sources.

The World Bank finds that intangible capital makes up 80% of total wealth, a finding that according to World Bank (2006) is characteristic of high income OECD countries. World Bank (2006) found an even higher estimate of 84% for intangible capital for the year 2000 for the Netherlands. Now it is remarkable that when we follow through the World Bank's residual approach using available official statistics, intangible capital in 2005 would only have a 59% share of total wealth (see Table 5.3), a figure that according to the World Bank is characteristic of low income countries. When we use direct estimates for intangible assets, this value increases towards 66% (see Table 5.4) of total wealth, which is still a lot less than the World Bank's estimate. The latter figure is according to the World Bank characteristic of middle income countries. The difference is primarily driven by the much higher values of urban land and produced assets, a topic to which we will return in the Discussion.

Sensitivity analysis

The value of total wealth calculated as the right hand side of Eq. 1 is highly sensitive to the assumptions used, such as the choice for a pure rate of time preference and time horizon, as shown in Table 5.5. The effect on the value of intangible capital calculated as a residual would be equivalent.

Table 5.5 Sensitivity analysis of wealth estimates for the Netherlands (2005)

Approach	Consumption	Horizon	Discount rate	Wealth	Return to wealth
		years	%	billion €	%
Residual	C_sna	25	1.5	7,802	5.3
Residual	C_sna	42	1.5	11,671	3.5
Residual	C_mew	25	1.5	7,125	5.8
Residual	C_mew - Educ	25	1.5	7,079	5.8
Residual	C_sna	25	3	6,557	6.3
Residual	C_sna	25	0	9,414	4.4
Direct				8,713	4.7

Source: Author's calculations.

The choice of a pure rate of time preference remains controversial. Table 5.5 provides results for two alternative scenarios: 0% (a very small number was argued by the Stern report in the context of climate change); 3%. The effect on 2005 estimates of wealth would be about 20%, a lot higher than the effect of using C_{MEW} rather than C_{SNA}, which as we saw in Section 5.3 is about 10%. The effect of subtracting defensive expenditures is relatively small.

The default time horizon over which benefits accrue used by the World Bank is 25 years. The reason provided is that this "roughly corresponds to a generation" (World Bank, 2011, p.143). An alternative line of reasoning may be to argue that total wealth should be estimated as the discounted consumption of a representative inhabitant. This could be made operational by taking the current life expectancy of an individual with an age equal to the current average age of the population at that time. For the Netherlands the average age of the population increases from 37.4 to 39 years in the time frame 1995–2005. The remaining life expectancy of a representative inhabitant of that age, remains fairly constant at 42 years, a lot higher than the 25 year default period. Choosing a time horizon of 42 years would raise the year 2005 total wealth estimate by about 50%. As expected, a change in the time horizon, due to the low discount rate used, has a very large effect on wealth estimates.

However, the fact that according to Eq. 4 net income equals the return to wealth constrains the magnitude of our total wealth estimate. The last column of Table 5.5 shows the implied returns for the wealth estimates of the sensitivity analysis. According to the World Bank (2012, p.93) "a 'normal' rate of return on assets should be on the order of 5%". From this perspective, it can be seen that our direct wealth estimate with a return of 4.7% seems reasonable.

5.4 Discussion

Our review of World Bank estimates of wealth for the Netherlands has demonstrated that there are large differences between empirical estimates of various types of assets, most notably urban land, produced assets, but for instance also regarding financial capital. Sometimes, these differences are due to the use of different data sources, but more often this is explained by the use of generic assumptions by the World Bank, which may be necessary given their objective to assess wealth for a large number of countries including developing countries for which official statistics on asset values are often lacking. Nevertheless, our research also points to a number of issues inherent in the World Bank's approach that may have wider ramifications for wealth estimates for other countries and therefore also for their general results.

First of all, as we saw in Section 5.3, the World Bank estimates the value of urban land as a fixed proportion of the value of fixed assets, for all countries, based upon an outdated source, which refers to data from Canada at best from 1984. To be fair, the World Bank (2006, p.147) acknowledges that "ideally, this proportion would be country-specific". However, there is an additional issue with this assumption, which is that it is unlikely that this proportion would remain constant even for a single country over the course of such a long time period. Indeed, by subtracting data on the value of agricultural land from the total value of land (both available from Statistics Canada, CANSIM database[13)]) we found that the ratio of urban land to produced assets gradually increased from 0.25 in 1985 towards 0.40 in 2005. It therefore appears that 24% is a conservative estimate. Second, although there are few countries that compile estimates for the value of land, Table 5.6 demonstrates that for those countries for which estimates are available the ratio of urban land to fixed assets is very different ranging from a mere 4% for the Czech Republic towards almost 80% in case of Australia and France. This may not come as a surprise, given for example differences in population densities.

Table 5.6 Comparison of values of fixed assets and land for selected countries (2011)

	Australia	France	Netherlands	Canada	Czech Republic
	billion dollars	**billion euros**	**billion euros**	**billion dollars**	**billion CZK**
	current 2010/11	2011, base 2005	current 2011	current 2011	current 2011
Rural land	265	436	87	210	370
Urban land	3,521	5,163	1,017	1,898	593
Fixed assets	4,439	6,648	1,990	4,038	13,722
	%				
Ratio (urban land / fixed assets)	79	78	51	47	4

Multiple sources: www.abs.gov.au - 1301.0 - Year Book Australia, 2012, Table 2.37; statline.cbs.nl - Table 'Inkomens- en vermogensrekeningen; niet-financiële balansen'; www.bdm.insee.fr - Annual National Accounts (base 2005) - Stocks and change in non financial assets by institutional sector; Ondrus (2011); Canada – see previous footnote (websites accessed June 2013).

13) We used CANSIM Table 002-0020 Balance sheet of the agricultural sector which provides information on the value of agricultural land from 1981 onwards and Table 378-0049 National balance sheet, which provides data on the total value of land and produced assets from 1970 onwards (both accessed Nov. 2012).

As a result, the value of intangible capital estimated by the World Bank is likely to be impacted for a number of countries, which may distort the comparison of intangible wealth over time and across countries.[14] A related technical issue is that the compilation of wealth estimates in constant prices requires that produced assets and land need separate deflators. Improving the estimation method for urban land, by allowing for country specific and time-dependent estimates, is of critical importance to the comprehensive wealth approach.

Second, although as we have demonstrated, there is a consistent theoretical framework that unifies ANS and comprehensive wealth accounting, empirical estimates of on the one hand ANS and on the other hand wealth are at risk of being inconsistent. First and foremost this is shown by the fact that the World Bank uses C_{SNA} data rather than C_{MEW} data that the underlying model implies for its compilation of wealth estimates. But an inconsistency also arises as soon as the estimates for produced capital (fixed assets) differ from the national accounts data, as the World Bank relies upon the latter for the consumption of fixed capital estimates (depreciation) which is an element of the ANS indicator (as shown for the Netherlands in Table 5.1, the ratio is 100%).

Third, regarding the estimation of ANS, degradation estimates by the World Bank appear to be conservative due to the limited number of pollutants that is included. As we saw in Section 5.3, for countries with a high population density as the Netherlands, the effect may be important, as the pollutants that are out of scope often impact local air quality. Our finding is consistent with Ferreira and Moro (2011) who found particularly high degradation values for SO_2, which was one of the main reasons they found negative ANS estimates for some years in Ireland, an issue that will be further investigated in Section 5.5. The use of comprehensive country-specific data on externalities, as recently estimated for instance within the EXIOPOL project[15] for a large number of countries and pollutants, may be an improvement.

The outcome of the residual approach is as we have seen highly sensitive to the assumptions used such as the discount rate and time horizon, although the interpretation of income as a rate of return to wealth does provide a constraint. Therefore obtaining direct estimates for human capital and other intangible assets is to be preferred from a statistical perspective. In the area of human capital

[14] The effect on the intangible estimate for the Netherland is about 8%, while the effect for Canada is about 3%.

[15] EXIOPOL is an acronym that stands for "A new environmental accounting framework using externality data and input-output tools for policy analysis" (www.feem-project.net/exiopol). The project has integrated research on externalities within an environmentally extended IO framework. The focus is on EU countries with additional areas in order to obtain world coverage for a diverse set of impacts.

measurement, advances have been made in recent years. The Jorgenson–Fraumeni approach has emerged as the preferred method in the statistical community (see for instance Wei, 2008), although a complete integration of human capital in national accounts would require a fundamental overhaul of the system (Aulin-Ahmavaara, 2004). As we have seen, data availability on direct human capital estimates is improving (OECD, 2011c). There have been also advances in estimating values for other intangible assets. With the acceptance of the 2008 SNA, expenditures for R&D – called intellectual property products (IPP) – will become capitalized (i.e. they are no longer regarded as intermediate consumption but as investments) and generally available. Drawing upon data sources that provide direct estimates of the intangible class of assets is a promising route to further improve wealth accounts.

5.5 Comments on Ferreira and Moro 'Constructing genuine savings indicators for Ireland, 1995–2005'

Ferreira and Moro's article (2011) is an important attempt to improve and expand the 'rough' World Bank genuine savings estimates of Ireland, based upon available official statistical sources of Ireland. Their main findings are that Irish genuine savings estimates are smaller than the Bank's estimates and – this may come as a surprise – even negative for a number of years. These results are primarily driven by their much larger estimates of environmental degradation. Unfortunately, when going over their calculations, I have come to doubt the main outcomes of their study due to two issues: CO_2 damage values are overestimated by a factor 13 as a mistake is made in converting damage costs expressed in dollars per ton carbon towards dollars per ton CO_2. Second, the implied average SO_2 damage values per ton that they use are much higher than found in the literature.

CO_2 damages overestimated by factor 13

Ferreira and Moro make a mistake when converting damage costs (global social costs) expressed in $/tC (i.e. US dollars per ton emitted carbon) into damage costs expressed in $/t $CO_{2,}$. This conversion is necessary as data on air emissions are usually expressed in tCO_2. Now, using the respective atomic masses, 1 ton of carbon (C) is equivalent to 44/12 = 3.67 tons of CO_2. This implies however, that when converting damage costs expressed in tC to damage costs in tCO_2 one should *divide* by 3.67 instead of *multiply* by 3.67 (CE, 2010). The intuition is that an emitted ton CO_2 contains only 12/44 ton of harmful carbon. However, Ferreira and Moro multiply, as they state on p.546 "The global social cost of CO_2 emissions used in our study, 14$/tC (equivalent to 51.33$/t CO_2)." The error this introduces in their results is about a factor 13.4 (3.67*3.67).

As shown in Table 5.7, this is not simply a small mistake in the write-up, but is also part of their estimates. Ferreira and Moro's results are roughly about a factor 10/11 higher than the estimates that can be obtained when using the same assumptions and data sources as Ferreira and Moro. The reason this factor differs from 13.4 could be due to the use of different ways of calculating constant prices in Euros and/or different vintage of emissions data. Table 5.7 also includes recent World Bank (2011) estimates as a further benchmark. The Bank's estimates are higher than my own estimates as they use 20$/tC instead of 14$/tC, but apart from that are in the same range. The consequence therefore is that Ferreira and Moro overestimate the CO_2 damage values by roughly a factor of 13.4.

Table 5.7 Irish CO_2 damages (2000-2005)

	2000	2001	2002	2003	2004	2005
	1,000 tonnes					
CO_2 emissions	44,100	46,310	44,708	43,760	44,402	45,920
	million 2000 €					
World Bank	271	288	285	279	281	281
Author's estimate	206	217	209	205	208	215
Ferreira/Moro (Fig. C1 - method 1) approx.	2,000	2,250	2,350	2,250	2,150	2,150
Ratio (Ferreira - Moro/our estimates)	9.7	10.4	11.2	11.0	10.4	10.0

Data sources: CSO 2010; DNB; Ferreira and Moro (2011); World Bank (2011). Estimates by Ferreira and Moro are read from their Fig. C1 and therefore not very precise. My estimates use the same assumptions and data sources as Ferreira and Moro (2011) i.e. a damage value of 14$/tC (assumed constant in constant prices) and official statistics on air emissions from the Irish Central Statistics Office. World Bank estimates (which are available in current $ prices) were first converted into constant year 2000 $ using US inflation rates; subsequently they were converted into 2000 € using the year 2000 exchange rate.

SO_2 damages overestimated

A second issue concerns the estimate of SO_2 damage values by Ferreira and Moro (2011) which amount to about 8,500 million € (see their Fig. 3 and my estimates in Table 5.8).

Table 5.8 Environmental degradation in Ireland (2000)

	Unit	
SO_2 emissions	**1,000 tonnes**	141
NO_x emissions	**1,000 tonnes**	142
PM_{10} damages	**million €**	79
NO_x damages	**million €**	398
Total environmental degradation	**million €**	11,000
SO_2 damages	**million €**	8,524
SO_2 damages	**1,000 €/ton**	61

Sources: PM_{10} (taken from World Bank, 2011, converted into €); NO_x 2800€ per ton NO_x; CO_2 - damages (Ferreira and Moro, Fig. C.1); Total (Ferreira and Moro, Fig. 3 – read from the graph); NO_x and SO_2 emissions (CSO, 2010).

As demonstrated in Table 5.8, dividing these SO_2 damages by the actual SO_2 emissions in 2000 would imply average SO_2 damage values per emitted ton SO_2 of about 61,000 €. Although Ferreira and Moro present in their paragraph 4.2 a very useful sensitivity analysis of their SO_2 (and NO_x) damage estimates, they do not mention explicitly that such a large average damage estimate per ton is *de facto* used in their calculations. Ferreira and Moro (p.547) refer to a study by Holland and Watkiss (2002) as a source for their calculations, who provide country specific damage values for the year 2000, distinguishing between emissions in rural and urban areas. As mentioned by Ferreira and Moro, for Ireland these values lie between 2,600€/ton SO_2 emitted in rural areas towards 45,000 €/ton for emissions in the most densely populated city Dublin. However, Ferreira and Moro do not explain how their large average damage estimate per ton can be obtained from these much lower source data.

To put these SO_2 damage values in perspective, the more recent AEA report from 2005 (which Ferreira and Moro refer to in another place, and which includes the same Holland and Watkiss as co-authors) provides marginal SO_2 damage estimates for Ireland for the year 2010 in the range of €4,800 – €14,000 (depending on whether VOLY or VOSL, median or mean values are used), indeed much lower (and even more so when these values would be expressed in year 2000 prices).

As a result of the above two issues, I have reason to believe that the environmental degradation costs are severely overestimated, which impacts the genuine savings estimates presented in the article. As degradation estimates are the key driver of the results, the main outcomes such as negative genuine savings for some years of their study seem to be undermined.

5.6 Conclusions

The objective of this chapter has been to review and refine World Bank wealth and ANS estimates for the Netherlands based upon a confrontation with country specific official data sources. The chapter also investigated the results of a similar study (Ferreira and Moro, 2011), who made a comparison between World Bank estimates and official statistics of the genuine savings indicator for Ireland.

We have shown that some of the 'stylized facts' (World Bank, 2011, p.6) concerning intangible capital (e.g. fastest growing; constitutes around 80% of total wealth for high income OECD countries are not corroborated for the Netherlands when confronted with official statistics.

Several directions for future improvement of wealth and ANS estimates were suggested. These pertain especially to obtaining country specific estimates for urban land values, as well as using a broader scope of pollutants in measuring degradation costs (a point also made by Ferreira and Moro, 2011).

In the near future, more research is clearly needed on estimating human capital directly as human capital constitutes such a significant part of wealth and because using the residual approach is highly sensitive to assumptions used. As exemplified by the discussion between the inclusive and comprehensive wealth approach (UNU-IHDP and UNEP, 2012; Hamilton, 2012), an important issue for wealth accounting will also be the choice of the asset boundary. A related issue that will be important to clarify is how wealth estimates are related to national accounts data on balance sheets.

As we have seen in Chapter 2 (World Bank, 2011, Chapter 8), wealth accounting is clearly a growing area. Country experiences are increasing in several areas, although the lack of official statistics for many countries continues to be an issue that needs further scrutiny.

5.7 Appendix

The basic model that underlies the definition of adjusted net savings is presented in Hamilton and Clemens (1999), whose exposition we follow here. Point of departure is a simple closed economy were a single resource (here we will assume it is non-renewable) is used denoted by q for producing a composite good. This good is either consumed (with C denoting consumption), invested (in produced assets K or human capital H), or used for pollution abatement (with a the corresponding expenditure) i.e. $F(K,q,H) = C + \dot{K} + a + m$. Education expenditure is transformed into human capital by the function $t(m)$.[16] The utility function is based upon both consumption as well as the existence of environmental services B. The environmental services B are negatively impacted by the existence of a pollutant stock X according to $B = B_0 - \beta X$. X increases as a result of emissions e which are considered to depend on both the level of production and abatement expenditure i.e. $e = e(F,a)$; naturally occurring dissipation d reduces the pollution stock. Hence, we obtain the following problem for maximizing wealth W:

max. $W = \int_t^{\infty} U(C,B)e^{-rs}ds$ subject to

$$
\begin{aligned}
\dot{K} &= F - \delta K - C - a - m \\
e &= e(F,a) \\
\dot{X} &= e - d \\
\dot{S} &= -q \\
\dot{H} &= t(m) \\
B &= B_0 - \beta X
\end{aligned}
$$

This yields the following expressions for the Hamiltonian:

$$M = U(C,B) + \gamma_1 \dot{K} + \gamma_2 \dot{X} + \gamma_3 \dot{S} + \gamma_4 \dot{H}$$

[16] In the context of this model H is restricted to human capital, rather than the broader category of intangible capital.

The control variables are $C,\ q,\ a$ and m. After derivation of the four necessary conditions (assuming that we have an interior solution); linearizing the utility function, and assuming that optimality holds (noted by *) we derive:

$$M^* = U_C C + U_B B + U_C \left(F - \delta K - C - a - m\right) + \frac{\gamma_1}{e_a}\left(e-d\right) - \gamma_1 (1 + \frac{e_F}{e_a})F_q q + \frac{\gamma_1}{t_m}\dot{H}$$

Dividing by U_C transforms the expressions for the Hamiltonian in utils into an expression in money units that is more easily comparable to economic statistics. This measure is termed measurement of economic welfare (MEW) in Hamilton (1996).

$$MEW^* = C + \dot{K} - \delta K + B + \frac{1}{e_a}\left(e-d\right) - (1 + \frac{e_F}{e_a})F_q q + \frac{t}{t_m}$$

Hamilton and Clemens (1999) argue that when considering carbon dioxide in practice it can be assumed that d (dissipation) is small compared to e (emissions), as its residency time is in the order of 200 years. The term $\frac{e_F}{e_a}$ is considered to be close to 0. As a result we obtain the following expression for economic welfare (in case of the assumed optimality):

$$MEW^* = C + \dot{K} - \delta K + B - D_E - D_N + A_H \qquad \text{(A1)}$$

with as shorthand

$$A_H \equiv \frac{t}{t_m}$$

$$D_N \equiv F_q q$$

$$D_E \equiv \frac{1}{e_a}\left(e-d\right)$$

Hamilton and Clemens (1999) compare the expression for MEW* with NNP as it is conventionally measured in the national accounts in order to see what a 'green NNP' would look like.

$$NNP = C + \dot{K} - \delta K \qquad \text{(A2)}$$

The first 3 items of Eq. A1 are equal to NNP (where we suppressed international trade for ease of exposition). However, we should be very careful in identifying C in the expression for MEW* with C in expression for NNP. Current (as opposed to capital) educational expenditure is part of government final consumption and included in C according to the SNA. However in the expression for MEW investment in education m is separately identified. Hamilton (1996) argues that current education expenditures m can be seen as an lower-bound estimate of investment in human capital $\frac{t}{t_m}$. Similarly, defensive expenditures by governments and households a are included in C according to the SNA. We can therefore write:

$$NNP = C_{MEW} + m + a + \dot{K} - \delta K$$

Next, as Hamilton (1996) argues, as NNP is a measure of production and not necessarily of welfare, environmental services represented by the term B are left out of consideration. We then obtain for a green measure of NNP:

$$NNP_{green} = C_{MEW} + ANS \tag{A3}$$

with

$$ANS \equiv GS - D_K - D_N - D_E + A_H \tag{A4}$$

where GS are traditional savings $\dot{K}$ and D_K equals δK. This results in the standard interpretation of ANS (in the context of this model) that corresponds to (net) investment in produced assets plus human capital minus the depletion of resources and value of degradation.

Derivation that C in Eq. 1 is C_{MEW}

We now examine the relation between ANS and wealth following Pezzey (2003). First we write the Hamiltonian as a function of all the variables it depends upon i.e. $M\left(K, S, X, H, \gamma_1, \gamma_2, \gamma_3, \gamma_4\right)$ and differentiate with respect to time. We obtain

$$\frac{\partial M^*}{\partial t} = \frac{\partial M^*}{\partial K}\dot{K} + \frac{\partial M^*}{\partial S}\dot{S} + \ldots$$

We can then use the equations of motion to derive that:

$$\frac{\partial M^*}{\partial t} = r(\gamma_1 \dot{K} + \gamma_2 \dot{S} + ..) \tag{A5}$$

As we know that $NNP_{green} = \frac{M^*}{U_C}$ we obtain for the time derivative that $\frac{dNNP_{green}}{dt} = \frac{\dot{M}^*}{U_C}$ as the utility function is autonomous and does not depend explicitly on time. We can then use Eq. A5 together with Eq. A3 to derive the following differential equation:

$$\frac{dNNP_{green}}{dt} = r(NNP_{green} - C_{MEW})$$

The solution of this differential equation yields that

$$NNP_{green} = r\int_t^{\infty} C_{MEW}(s)e^{-r(s-t)}ds \equiv rW_t$$

Therefore C in Eq.1 is equal to C_{MEW}

6. Analysis of changes in Dutch emission trade balance(s) between 1996 and 2007

6.1 Introduction[1)]

In a globalizing world economy which exerts increasing pressures on the natural environment, an important policy question is how the responsibility for pollution can be allocated to national economies. There are several frameworks for estimating a country's greenhouse gas emissions yielding different results (Statistics Netherlands, 2010b). First, the IPCC (Intergovernmental Panel on Climate Change) has drawn up specific guidelines to estimate and report on national inventories of *anthropogenic* greenhouse gas emissions and removals (IPCC, 1996). Second, Statistics Netherlands annually publishes the greenhouse gas emissions that actually take place on the Dutch *territory*. Third, Statistics Netherlands also annually publishes the total greenhouse gas emissions by economic activities, often termed a *production* approach, which are calculated according to national accounting principles, as part of the environmental accounts (UN et al., 2003). These three frameworks serve different purposes and accordingly differ in their treatment of emissions caused by international transport, tourism, biomass combustion, and carbon sinks.

A complementary fourth approach is the *consumption approach* in which emissions required for the satisfaction of consumption requirements of countries are estimated, using environmentally extended input-output analysis (E-IO). This entails that emissions embodied in imports are included, while emissions inherent in exports are excluded. There is a growing interest in the compilation of consumption-based indicators, as evidenced by the recent proposal to include demand-based emissions as one of the headline indicators of the OECD's green growth strategy (OECD, 2011a). A related indicator is the difference between consumption based and production based emissions which is usually called the emission trade balance (ETB) (De Haan, 2004). ETBs can be compiled both on a macro and bilateral basis. A positive ETB indicates that countries pollute more for others, while a negative balance indicates that foreign countries pollute in order to satisfy domestic consumption needs. The compilation of ETBs therefore provides important information for (inter)national climate policy.

Several statistical offices started experimenting with E-IO techniques in order to estimate consumption based emissions in the mid 1990s (Canada, Denmark, and the Netherlands) later followed by others (e.g. Germany (Destatis, 2010), Sweden,

[1)] This paper is based upon Edens et al. (2011) (with only minor changes). We would like to acknowledge Cees Withagen for his comments and Glen Peters for fruitful discussions on typologies of embodied emissions.

and France). Estimates were obtained usually based upon so-called Single Region Input Output models. These estimates often (with the exception perhaps of Canada (Statistics Canada, 2006)) lack the status of official statistics and are conceived as the results of analysis or pilot projects. In the research community, there is a vast and quickly expanding literature on the subject (see Wiedmann, 2009; Hoekstra, 2010 for overviews) that often use comprehensive Multi-Regional Input-Output (MRIO) models. Due to their significant data requirements as well as reliance on foreign data sources, these models have been outside the scope of statistical offices. For the same reasons there have been relatively few MRIO based studies that compile time series of carbon footprints.[2)]

The purpose of this Chapter is to analyze changes in the Dutch emission trade balance(s) – both macro and bilateral – between 1996 and 2007, based upon several possible decompositions. In this study we use the method developed by De Haan (2004) to obtain a cross section of the country specific ETBs. This allows us to decompose each bilateral balance into a composition, volume and technology effect. Second, we will perform a structural decomposition analysis (SDA) of embodied import and export emissions separately to analyze the driving forces behind changes in the macro ETB over time. Finally, we also distinguish between three different greenhouse gases CO_2, N_2O, and CH_4. The results are important for assessing the effectiveness of current climate policies in curbing emissions.

For our purposes, following the typology introduced by Peters (2008), we follow an Emissions Embodied in Bilateral Trade (EEBT) approach in defining import and exports emissions, in contrast to an MRIO approach. According to EEBT, the import emissions of country A correspond to the export emissions of country B. This method does not keep track of whether imports via intermediate consumption are turned into goods which may again be exported. The supply chains are deliberately kept short and focus only on individual trade between countries. As Peters notes, neither method is correct or incorrect, but they do result in different allocations of import and export emissions across countries.

There are several reasons why we use an EEBT approach. The EEBT approach is comparable to bilateral trade data. This facilitates the use of cross-sectional analysis and allows us to interpret changes in the balance over time more easily. It is also of a lower complexity, and in particular less-data demanding than a full MRIO model.

[2)] A recent exception is Wiedmann (2010).

The outline of our paper is as follows: we will commence with a brief exposition of the EEBT model, the cross-sectional and SDA forms. In Section 6.3 we will discuss data sources and several conceptual issues such as treatment of non-competitive imports; trade and transport margins and re-exports. In Section 6.4 we will present results followed by a discussion in Section 6.5 and conclusions in Section 6.6.

6.2 Model, cross-sectional and decomposition analyses

EEBT

The embodied emissions in an EEBT model can be expressed (Peters, 2008) as follows:

$$f_g^{rs} = b_g^r (I - A^{rr})^{-1} e^{rs} \tag{1}$$

where:

f_g^{rs} the embodied emissions in exports from region r to s of green house gas type g (we distinguish CO_2, N_2O, CH_4 and total GHGs);

b_g^r the vector of emission coefficients of country r for greenhouse gas g;

A^{rr} the technical coefficients matrix for the domestic economy r;

e^{rs} export vector from region r to region s;

Likewise, the import emissions can be expressed as

$$o_g^{sr} = b_g^s (I - A^{ss})^{-1} e^{sr} \tag{2}$$

where:

o_g^{sr} the vector of embodied emissions in imports from region s to r of greenhouse gas type g.

Cross-sectional analysis

If we make the domestic technology assumption that the foreign IO tables can be replaced by the Dutch domestic IO table and that the emission intensities are the same, we can derive that the bilateral ETBs can be expressed as (where 1 denotes the Netherlands):

$$ETB_g^s = b_g^1 (I - A^{11})^{-1} (e^{1s} - e^{s1}) \tag{3}$$

It consists of the difference between the export and import vector pre multiplied by the Leontief inverse of the domestic IO table.

The difference between the import and export vector can be decomposed into a volume and composition effect as shown by De Haan (2004). De Haan (2004) is – to the best of our knowledge – one of the first authors to use a cross-sectional analysis of the ETB. His technique was however based upon a so-called gross method of recording.[3)]

First we express the shares of industries in imports and exports respectively as:

$$\tilde{e}_k^{s1} = e_k^{s1} / \hat{e}^{s1}$$

$$\tilde{e}_k^{1s} = e_k^{1s} / \hat{e}^{1s}$$

where the breakdown by industry k is now made explicit in the notation of the export and import vectors. Now $\hat{e}^{1s} = \sum_k e_k^{1s}$ denotes the total exports in monetary terms from the domestic economy to country s. A similar notation is used for the import vector. Herewith we can decompose the bilateral trade balance (Eq. 3) into a volume and a composition effect as follows:

$$\begin{aligned} ETB^s &= VOL^s + COM^s \\ VOL_g^s &= b_g^1 (I - A^{11})^{-1} \left[(\tilde{e}_k^{1s} + \tilde{e}_k^{s1}) * (\hat{e}^{1s} - \hat{e}^{s1}) \right] / 2 \\ COM_g^s &= b_g^1 (I - A^{11})^{-1} \left[(\tilde{e}_k^{1s} - \tilde{e}_k^{s1}) * (\hat{e}^{1s} + \hat{e}^{s1}) \right] / 2 \end{aligned} \tag{4}$$

The intuition behind this decomposition is that the volume effect gathers all terms which express the difference between total exports and imports, whereas the composition effect gathers terms that take the difference between industrial shares. Eq. 4 is based upon the domestic technology assumption. In order to isolate

3) The distinction between MRIO and EEBT resembles the distinction between gross and net recording of embodied emissions introduced in De Haan (2004) but is on closer inspection slightly different. According to De Haan, whose analysis was based upon the two country pollution model, the difference between a gross and net method of recording consists in the treatment of emissions that via intermediate consumption are assigned to exports. These emissions would be included in both import and export emissions under a gross recording, but excluded under a net recording. As a result, the macro ETB under a gross and net recording would be equal, but the country specific ETBs would be different. In De Haan (2004) an additional distinction between gross and net method of recording is that re-exports are included in a gross method of recording, but not in a net method of recording.

also the effect caused by differences between bilateral emission intensities, we follow a two step approach: first the decomposition (Eq. 4) is calculated based upon the domestic technology assumption; second the bilateral ETBs with country (and industry specific) intensities are calculated (i.e. difference between the outcomes of Eqs. 1 and 2. The difference between both calculations gives us exactly the technology effect i.e. the part of the bilateral ETB that is caused by differences in bilateral emission intensities.

Structural decomposition analysis

Where a cross-sectional analysis provides a breakdown of the bilateral ETBs in a specific year, SDA is a technique used to changes in variables over time into its underlying causes. Based upon standard methods for SDA (De Haan, 2001) it is common to decompose changes in emissions over time into various effects (e.g. changing emission intensities – or fuel mix; changing structure of final demand; and volume of final demand). Although SDAs are usually applied to decompose total production based emissions, here we apply SDA to analyze changes separately for embodied export and embodied import emissions. Instead of using the total final demand vector as our volume term, we will only use the export (and import) vector.

$$f^{1s} = \underbrace{b^{1}}_{techno\log y} \underbrace{(I - A^{11})^{-1}}_{structure} \underbrace{\tilde{e}^{1s}}_{composition} \underbrace{\hat{e}^{1s}}_{volume} \tag{5}$$

In order to decompose the development of embodied export emissions, we use Eq. 1 to estimate embodied export emissions in 1996 and 2007. We then use the right hand side of Eq. 5 to decompose the difference into four factors: technology; structure; composition of export; and volume of exports. We focus here on decomposing developments in macro totals and do not further decompose embodied export or import emissions by region of origin (import) or destination (export). Our weighting method is based upon Dietzenbacher and Los (1998) (in this case we have 4! or 24 different equations), and we use a chained series of IO tables, where each IO table is expressed in both current prices and prices of the previous year (De Haan, 2001).

For decomposing the development of embodied import emissions, we also use a domestic technology assumption and calculate the development of embodied import emissions with Eq. 2. This implies that we are allowed to use Eq. 6 as our decomposition:

$$o^{s1} = \underbrace{b^1}_{techno\log y} \underbrace{(I - A^{11})^{-1}}_{structure} \underbrace{\tilde{e}^{s1}}_{composition} \underbrace{\hat{e}^{s1}}_{volume} \quad (6)$$

Subsequently we reconcile these domestic technology assumption based results with the outcomes we obtain for the development of embodied import emissions when using country specific emission intensities (Eq.2). The difference is added to the technology effect.

6.3 Data sources and preparation

As the purpose of our paper is to compare changes over time, it is essential to estimate ETBs in a consistent manner, based upon the same data sources and methodology. The choice for 1996 and 2007 is due to the fact that 2007 is the most recent year for which a definitive IO table is available, while 1996 is the first year in which trade statistics is digitally available in sufficient detail. The main objectives of our methodology are to ensure consistency with national accounts data and principles as well as with data from the environmental accounts.

The treatment of re-exports

The Netherlands is a small open economy that is characterized by large transit trade as well as re-exports.[4)] More than 50% of Dutch manufacturing exports consist of re-exports (Mellens et al., 2007) which makes it by far the largest European re-exporter (e.g. in Germany this lies around 15%). Re-exports are usually corrected for when MRIO models are used (e.g. Global Trade Analysis Project – GTAP), however, re-exports are difficult to measure and may give rise to large uncertainties especially when they constitute such a large part of trade.

Our research has shown that international databases show large discrepancies compared to official Dutch statistics. For instance, the total Dutch imports in 1998 available in the UN Comtrade database[5)] differs as much as 20% compared with

4) Re-exports are goods that are transported via the Netherlands during which they are (temporarily) owned by a resident and that do not undergo significant industrial processing. When no change of ownership occurs we are dealing with transit trade.
5) comtrade.un.org; accessed Jan 2010.

Statistics Netherlands' international trade statistics. There are also significant differences between the Dutch IO table (2004; current prices) and the IO table that is included in GTAP7. While export estimates are comparable (+1%), government consumption (+86%), investments (+50%) and import estimates (+23%) are much higher within GTAP7: the Netherlands is even portrayed as a net importer. These discrepancies could be due to a number of reasons such as the use of an outdated Dutch IO table and subsequent adaptations based on macro-economic developments and trade linking requirements. This implies that results for the Netherlands based upon GTAP7 data may have a strong bias and overestimate consumptive emissions (as reported for instance in Davis and Caldera, 2010; Hertwich and Peters, 2009).

For obvious reasons, we favour using official data sources for the Netherlands as much as possible. In the Netherlands, a lot of effort is put into separately identifying re-exports in trade statistics. This is done based upon a modelling approach in which for each establishment import and export data are compared. When imports and exports occur for an establishment in the same accounting period for similar products (defined on the basis of identical first 6 digit HS codes) they are classified as re-exports. The results of this exercise are then cross-checked with other data sources for validation. The subsequent integration of trade data into the national accounts provides additional possibilities to compare with other data sources such as production statistics and improve the estimates.

Disaggregating imports and exports of the IO table into regions

There are several conceptual differences between international trade statistics which in principle follow the cross-border principle and the national accounts that are based on the residence principle. These differences are for instance due to merchanting and goods sent abroad for processing. As a result different values for imports and (re)exports are reported in trade statistics and in the national accounts. However, the import and export data available from the national accounts do not provide sufficient country detail as they only distinguish between intra and extra EU. In our methodology we therefore first derive total import and export from the national accounts, while in a subsequent step we use international trade statistics data to distribute these vectors across regions. The following steps are taken:

- First an IO table is compiled of the industry by industry type [60 60] in current prices. To be precise, we distinguish in our IO table 59 activities and a pseudo industry "margins" (see next section). Although a higher level of disaggregation of the IO table exists (maximum [118 118]), we have chosen to bring the IO table to the level of detail available in our air emission accounts. This avoids the need to break down air emission data in greater industry detail, which would require additional assumptions.

- Import and export vectors e^{rs} are constructed in such a way as to assure consistency with national accounts data. This is achieved by using international trade statistics not for its levels, but only for calculating fractions of imports and exports for each imported product (goods and services). First, using international trade statistics on goods we calculate proportions for the 17 countries/regions[6)] that we distinguish in our model for each imported good. A similar procedure is followed for trade in services, after which we integrate both data sets. As a result we have import by industry (use) of each product disaggregated by region.
- In a final step, these import data on goods and services are allocated to supplying industries for each region individually. This is done by assuming that all trade partners have an identical production structure as the Netherlands. For instance, if a certain product X is produced by two different industries in proportion A/B than also the imported product X is assumed to be produced by these two industries in the same ratio. This results in an import vector per region.
- Likewise, also the exports are disaggregated into the 17 countries/regions by industry using similar procedures as for imports i.e. based upon international trade in goods and services data. However, in case of exports we know exactly which industries produce which export products. The result is a matrix of 17 countries/regions by 60 industries that decomposes the final demand vector of exports from the original IO table.

Emission coefficients

Emission coefficients have been estimated for the year 2007 as well as 1996 for 17 countries/regions, for the 60 industries that we distinguish in our air emission accounts, separately for CO_2, N_2O, and CH_4. The emission coefficients are defined as the total emissions divided by gross output in basic prices. As our IO table is in current prices, also the intensities are expressed in emissions divided by output in current prices.

6) These regions are: Germany; Belgium; USA; UK; China; France; Russian Federation; Italy; Spain; Japan; Sweden; Eastern Europe; other Western; Africa; South and Central America; other Asian; and the Middle East.

Regarding the emissions data, for most EU countries data from the air emissions accounts from Eurostat have been used. For the non-European countries the most important data sources are: IEA database; UNFCCC database; and data obtained from Wilting[7]. For some countries additional sources have been used such as data available on National Statistical Institutes' (NSI) websites. Sources used for the estimation of gross output are: UN data; and other sources (NSIs). Finally, when emissions and output data were available for different years, output data has been extrapolated based on inflation estimates.[8]

Conceptual issues

Treatment of non-competing imports

In the Dutch IO table, one of the largest non-competing imports consists of purchases by Dutch tourists abroad. However there are no direct data sources that would allow a breakdown of these expenditures of Dutch tourists abroad across industries as in general inbound tourism is better measured than outbound tourism. We assume therefore the following:

- Expenditures by Dutch residents abroad follow the same breakdown as the expenditures of foreign tourists in the Netherlands.
- Secondly, total expenditures are distributed over regions based on tourism statistics, specifically, based upon the number of days spent on average in a particular country.

Other non-competing imports are assigned to one single producing industry. For instance tobacco is assigned to agriculture.

Treatment of trade and transport margins

Trade and transport margins are separately recorded in an additional row and column in the Dutch IO table outside the intermediate demand block. The reason is that since the 1987 revision of the national accounts, all industries are functionally recorded i.e. including their secondary activities. Due to lack of accurate information regarding the destination of industry specific margins, the trade and transport margins that are produced as a secondary activity by a non trade industry are registered as an additional final demand category "margins". The total of the final demand column is distributed via an extra row "margins" outside the intermediate demand block across intermediate and final demand categories.

7) Personal correspondence.
8) The estimated emission coefficients have been checked by various experts.

Performing IO analysis therefore becomes more difficult as the standard intermediate block excludes these types of transport activities. The solution that we found (see also Peters and Hertwich, 2006 for a similar approach) is the following: When constructing the IO table the final demand column "margins" as well as the row "margins" are pulled inside the intermediate demand block creating a fictional industry "margins". The IO table therefore increases in size (+1). The emission intensity of this fictional industry is set equal to 0.

6.4 Results

Macro trade balance for individual greenhouse gases

The ETB for greenhouse gases (excluding Fluorinated gases) for the Netherlands with the rest of the world in 2007 was negative and amounted to −32 Mton CO_2 equivalents. Resident production emissions equalled 237 Mton CO_2 equivalents, whilst consumption based emissions amounted to 269 Mton. In 1996, the macro ETB was slightly positive at 4 Mton. This indicates that during this period the Netherlands has become a net importer of emissions.

Whereas the production emissions decreased by 4% between 1996 and 2007, the consumption emissions increased by 11%. The embodied import emissions have increased by 37% whereas export emissions increased by only 3%.

Table 6.1 Consumption and production emissions for different types of greenhouse gases[1] (1996 and 2007)

	Total GHG		CO_2		CH_4		N_2O	
	2007	1996	2007	1996	2007	1996	2007	1996
	Mton CO_2-equivalents							
1. Emissions embodied in imports	144	105	97	64	34	24	12	17
2. Emissions embodied in exports	112	109	93	85	8	9	11	16
3. Emission trade balance = 2–1	−32	4	−4	20	−26	−15	−1	−1
4. Emissions by residents (production approach[2])	237	246	204	201	17	21	16	24
5. Worldwide emissions for Dutch consumption needs[3] = 4–3	269	242	209	181	43	36	17	25

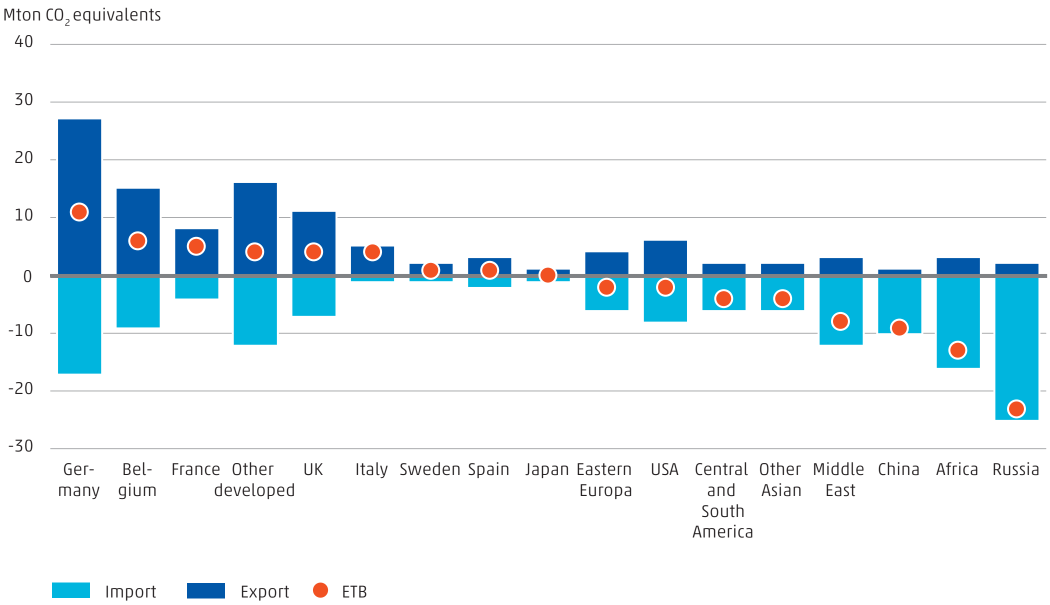
Sources for production emissions: Dutch air emission accounts (Statistics Netherlands, 2010b).
[1] Excluding Fluorinated gases.
[2] Including direct emissions by households.
[3] Consumption includes both intermediate and final consumption as well as direct emissions by households.

The emission trade balance can also be compiled separately for the three most important greenhouse gases: CO_2, CH_4 (methane) and N_2O (nitrous oxide). As Table 6.1 shows, the CO_2 balance has become slightly negative in 2007, while it was highly positive in 1996. The balances for nitrous oxide and, especially, methane have remained negative. The negative trade balance for methane has become increasingly negative between 1996 and 2007, while the negative balance for nitrous oxide has remained constant.

ETBs for individual regions and countries

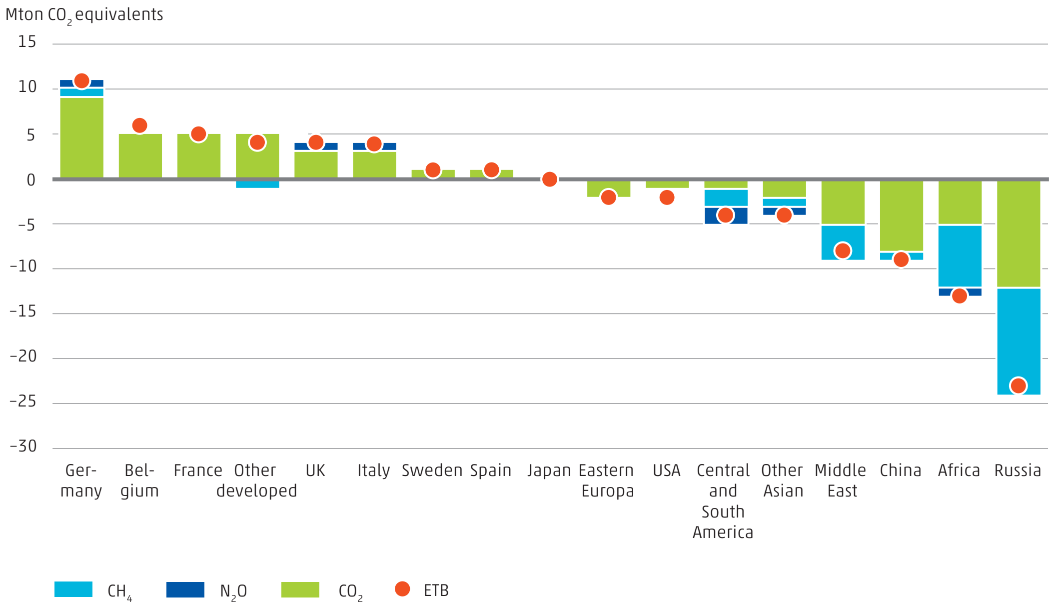
Figure 6.2 presents the results of the bilateral ETB balances of the Netherlands with 17 individual countries/regions. It is a consistent decomposition of the macro results. The bilateral balances are positive with OECD countries such as Germany, France and Belgium, but negative with developing and transition economies such as China, Russia, other Asia, and Africa.

Figure 6.2 Bilateral ETBs decomposed into import and export emissions (2007)



The decompositions by type of greenhouse gas as shown in Figure 6.3 provide insight in what is driving the region specific ETBs: in case of Germany we see that the positive ETB is primarily due to a positive CO_2 balance. With Russia we see that the balance is negative due to strongly negative methane and carbon dioxide balances. This is partly due to the fact that the Netherlands imports large quantities of oil and to a lesser extent natural gas from Russia. During its production, large amounts of CO_2 and CH_4 are released because of venting and flaring during extraction as well as leaks during transportation.

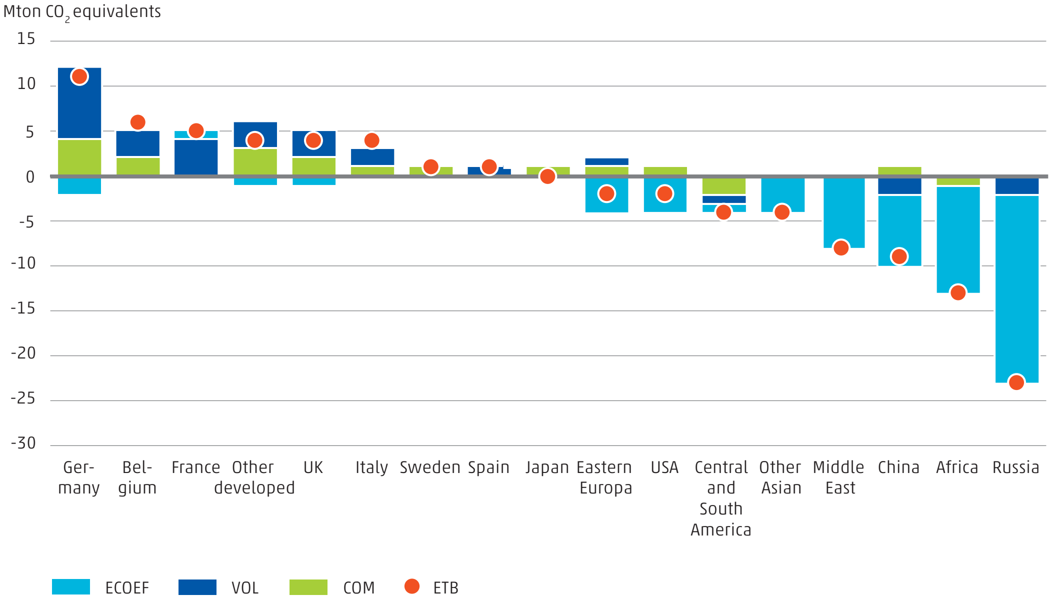
Figure 6.3 Bilateral ETBs decomposed into type of greenhouse gas (2007)



Cross-sectional analysis

In order to assess the driving forces behind individual ETBs, we perform a cross-sectional analysis as described in Section 6.3, in which the individual balances are broken down into three effects: a technology effect that measures differences in emission intensity between countries, a volume effect that measures the difference in value between imports and exports, and a composition effect that measures differences in the composition of imports and exports (see Figure 6.4).

Figure 6.4 Cross-sectional analysis of the bilateral ETBs (2007)



First of all we see that the choice of domestic coefficients instead of foreign has a significant impact on the results, especially for countries such as Russia, Africa and China whose cross-sections are dominated by a negative technology effect. By contrast, France and to a lesser extent Italy and Sweden have compared to the Netherlands on average cleaner technologies, as evidenced by the positive technology effect. In the case of France this is explained by the large use of nuclear energy. The macro ETB in 2007 which was negative at –32 Mton would, in the hypothetical case all Dutch bilateral trade partners would use Dutch technology (i.e. Dutch emission intensities), turn positive to 33 Mton CO_2 equivalents. The Netherlands can therefore be characterized as a country with relatively clean production technologies.

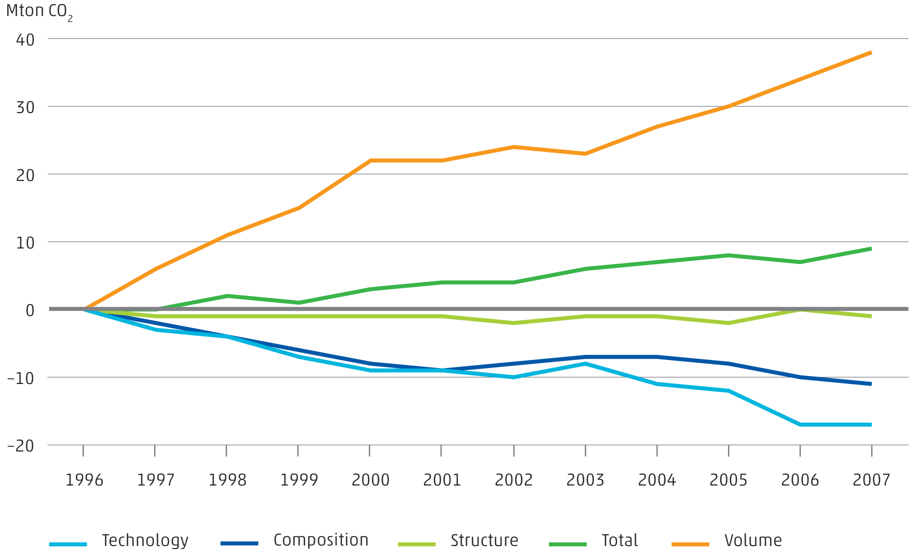
The composition effect which measures the extent to which the ETB is driven by differences in composition between imports and exports is positive for most regions which demonstrates that the Netherlands is a large exporter of emission-intensive products for instance from the chemical industry and horticulture. Exceptions are countries from which the Netherlands imports emission-intensive resources such as agricultural products (Central and South America and Africa – primarily driven by N_2O). The macro composition effect in 2007 is positive and contributes to a positive ETB balance.

The volume effect is positive with most regions which underlines that the Netherlands is a net exporter of products. The largest exceptions are China and Russia from which we import more than we export.

Comparison over time

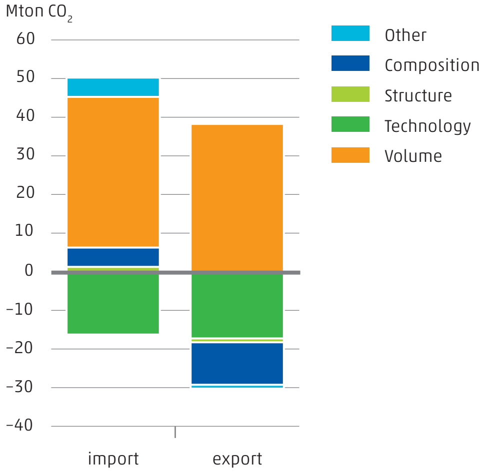
Figure 6.5 shows a SDA of CO_2 export emissions between 1996 and 2007, which according to Table 6.1 increased by 8 Mton.

Figure 6.5 SDA of export emissions (1996-2007)



It shows that growth in volume of exports – all else equal – would have resulted in export emissions being almost 40 Mton higher. This driving force was however countered by improved technology and a change in composition towards a cleaner mix of export products. For instance, the share in exports of the 10 most emission intensive industries in 1996, decreased by 1% compared to 2007, while the share of the 10 least intensive industries increased by 3%. To give some examples, the relative share in exports of horticultural products, which are emission intensive, decreased, while exports of emission extensive products such as post and telecom services increased. The structure effect which describes changes in the production structure of the Dutch economy, is not very significant during this period.

Figure 6.6 Cumulative structural decomposition results for imports and exports between 1996 and 2007



For the import emissions we only have estimates for foreign emission intensities for 1996 and 2007. Therefore, Figure 6.6 shows only cumulative results for 1996–2007.[9)] It shows that import emissions are also to a large extent driven by the volume effect, which is of similar magnitude compared to exports. The most striking difference is however due to the composition effect. While this is a negative driving force for exports, this is a positive driving force behind import emissions. What this demonstrates is that the Dutch economy increasingly exports clean products and imports dirty products.

This is consistent with the findings of the cross-sectional analysis: although the macro composition effect was positive for the year 2007, the cross-sectional composition effect was more positive in 1996.

9) The results for the decompositions for exports and imports are not exact, which is shown in the figure as an additional category other. This is due to the procedures used for deflating, in our case the use of chained IO tables. As the consumption vector is deflated according to a GDP deflator, while exports are deflated according to the export deflator, their ratio differs slightly when expressed in current or constant prices. This causes the decomposition path for exports to deviate slightly. The sum of SDAs for separate final demand categories however still exactly equals the difference in NAMEA air emissions between 1996 and 2007.

6.5 Discussion

The strength of our methodology is that it is fully consistent with national accounts concepts such as residence and national accounts data; this requires the use of additional data sources such as tourism statistics that are – as far as we are aware of – normally not used in other studies. Also the integration of trade in goods statistics with trade in services statistics provides value added. We use data sources that are all available in year t-1 which allows us to compile preliminary estimates and definitive estimates (as presented here) for t-3. The estimation of embodied export emissions is highly accurate due to the exact match between the level of detail of the IO table and air emission accounts.

A disadvantage of our method is the low level of disaggregation of products when trade linking. Our method ensures that we isolate re-exports in a manner consistent with national accounts data. This implies however that the number of products that we are able to distinguish in our analysis is dictated by the level of disaggregation that is available in the database from which the IO table is derived, which is only around 228 separate product groups. This reduces the possibilities to identify non-competitive imports. Second, we still impose the Dutch IO table – but not Dutch emission intensities – as economic structure on trade partners. The impact this may have on the results is hard to indicate. Rørmose et al. (2009) found in case of Denmark in 2005 when instead of Danish economic structure region specific structures were used that imported CO_2 emissions increased by about 15% (using a unidirectional trade model). This issue is something we intend to address in future research.

It is important to emphasize that the results that we obtain are partly driven by the model used. For instance when we made preliminary estimates for the year 2009 based upon a unidirectional MRIO for estimating import emissions, as reported in Statistics Netherlands (2010b), we found that the import emissions were a lot lower.[10] This is partly because the Netherlands is a small open economy, with a large trade sector, and some of the import emissions that are included within an EEBT model, would be excluded in an MRIO model, as these are assigned via intermediate consumption behind exports to the countries that finally consume these products. However, the difference is also due to the fact that the volume of Dutch imports significantly decreased between 2007 and 2009 because of the financial and economic crisis.

[10] According to Statistics Netherlands, 2010b, the import emissions amounted to 99 Mton CO_2 equivalents in 2009.

6.6 Conclusions

While many studies have estimated the level of consumption based emissions of nations, it is equally important to account for changes over time in a consistent way. To draw an analogy, policy makers primarily are interested in the growth rate of GDP, rather than the absolute level of GDP. The point of departure of this study therefore has been to first calculate embodied import and export emissions for two different years in a consistent way. Second, to understand what causes changes in embodied import and export emissions – what we have called trade balances – over time.

To understand changes of trade balances over time, we have used various decompositions of these macro totals: by region; by type of greenhouse gas. Moreover, we performed a cross-sectional analysis of bilateral ETBs into a volume of trade, composition and technology effect (same year), and structural decomposition analyses for the development of import and export emissions over time. The main findings are that the Dutch emission trade balance has worsened over time due to a strong increase in the import emissions of primarily CO_2. This is caused by the changing composition of trade: the Dutch economy increasingly exports clean products and imports dirty products.

These results have clear policy significance for (inter)national climate policy. Our results show that import emissions have increased by 37% whereas export emissions increased by only 3%. These increases have occurred notwithstanding the strict compliance of the Dutch Government with the Kyoto protocol. It is therefore important to complement production based registrations with consumption based accounting.

For instance, Dutch crude oil imports have remained fairly constant between 2000 and 2010 at 53 million tons, but the composition of imports has changed drastically. While Saudi Arabia, the United Kingdom and Norway formed the top three in 2000, in 2010 around one third of crude oil is imported from the Russian Federation which has become the largest import country, while Nigeria became the fifth largest (Brummelkamp and Sardjoepersad, 2011), both countries with a poor environmental record. Results of the cross-sectional analysis could be used in identifying Dutch trade partners in terms of their environmental record. Shifts in the composition of imports can be monitored by keeping track of the composition effect and technology effect related to imports over time introduced in Section 6.4. A policy response could be not to import products from countries which are above average emission-intensive or to ensure that embodied emissions per product do not decrease as a result of substitution of imports towards environmentally unfriendly countries. In the end, this may help to diminish total emissions embodied in national consumption.

However, there has been little standardization so far in carbon accounting, and different definitions of consumption, or imports and export emissions can be found. Increasing policy interest as well as wide proliferation of websites that allow calculating carbon footprints raise the question what the role and responsibility of the statistical community is in estimating and publishing official consumption based estimates. This pertains not only to issues of standardization but also, of the robustness and reliability of estimates. With the advance of MRIO models developed in several research consortia (GTAP, EXIOPOL, WIOD etc.), also the current practices of statistical offices who regularly compile estimates of consumptive emissions are called into question. We believe that cooperation is absolutely necessary to ensure synergy between the expertise of both statisticians and researchers. The implementation of the 2008 SNA with revised guidelines for the recording of good sent abroad for processing (Van Rossum et al., 2010), and to a lesser extent, the technical revisions of classifications of products and industries, will have important ramifications for IO analysis. These changes further underline the necessity to enhance cooperation for instance through the set-up of an international expert working group.

7. Conclusions

Main results

The main objective of this thesis has been to reconcile theory and practice in green/environmental accounting, which led to the research question: what are the possibilities to narrow the gap between theory and practice in green/environmental accounting? The research question was addressed by investigating different domains: the classic case of the extraction of a non-renewable resource and how to estimate costs of depletion; the emerging area of ecosystem accounting; wealth accounting; and, an application of environmental accounting in the form of environmentally extended input-output analysis.

Chapter 3 reviewed a number of depletion measures that have been recently brought forward in the context of environmental accounting ('practice') and green accounting ('theory'): depletion as a change in total wealth; depletion as 'using up' of the resource; depletion as net savings; or, depletion as net investment. The differences in assumptions between these measures are clarified by contrasting their approaches with the classic theory of a firm engaged in extraction. All measures are evaluated using a time series of data on Dutch natural gas reserves. The main findings are that correcting for the cost of depletion would lead to significant adjustments of both level and growth rates of Dutch net national income, with a strong dependency on the chosen measure. The chapter counters criticism that accounting in practice would necessarily underestimate depletion as shown by a counter example. It is argued that the choice for a depletion measure should be determined by the context of use: measurement of social welfare or sustainable income. The physical measure put forward in the SEEA Central Framework can be justified by its consistency with the income concept that underlies the SNA, whose objective is to provide an aggregate measure of economic activity.

Chapter 4 identified four key methodological challenges in developing ecosystem accounts: the definition of ecosystem services in the context of accounting, their allocation to institutional sectors; the treatment of degradation and rehabilitation, and valuing ecosystem services consistent with SNA principles. The different perspectives taken on these challenges are analyzed and a number of proposals are presented to deal with the challenges in developing ecosystem accounts. These proposals comprise several novel aspects, including (i) presenting an accounting approach that recognizes that most ecosystems are strongly influenced by people and that ecosystem services depend on natural processes as well as human ecosystem management; and, (ii) recording ecosystem services as either contributions of a private land owner or as generated by a sector 'Ecosystems' depending on the type of ecosystem service. We also present a consistent approach for recording degradation, and for applying monetary valuation approaches in the context of accounting.

The World Bank (2011) recently published times series of comprehensive wealth and adjusted net savings (ANS) estimates for over 120 countries. Chapter 5 reviewed and refined these estimates for the Netherlands, by comparing them with official Dutch statistics. The main empirical findings are that our ANS estimates are 13% of gross national income compared to 15% according to the World Bank due to higher degradation costs as a wider range of pollutants is covered. We also find that that intangible capital constitutes a far smaller share of total wealth (59% when a residual approach is followed; 66% when using direct estimates) than found by the World Bank (around 80%). This can be explained by the use of generic assumptions by the World Bank in combination with the use of different data sources, which result in large differences in the valuation of urban land, produced and financial capital. Another reason is due to the fact that the World Bank includes educational expenditures in its derivation of total wealth, whereas according to the underlying model these should rather be considered as an investment in human capital. Several directions for future improvement of wealth and ANS estimates were suggested such as obtaining country specific estimates for urban land values, as well as using a broader scope of pollutants in measuring degradation costs (a point also made by Ferreira and Moro, 2011). The chapter also contains a critical review of Ferreira and Moro (2011) who have done a similar exercise for Ireland and it is argued that the environmental degradation costs which are the main driver of their negative ANS estimates are severely overestimated. In the near future, more research is clearly needed on estimating human capital directly as human capital constitutes such a significant part of wealth and because using the residual approach is highly sensitive to assumptions used concerning the discount rate and time horizon.

In Chapter 6 bilateral emission trade balances (ETBs) for The Netherlands are constructed with 17 countries/regions and the results are compared for 1996 and 2007 for three different greenhouse gases. We establish a cross-sectional analysis of bilateral ETBs into a volume of trade, composition and technology effect. In order to analyze the driving forces of changes over time we perform a structural decomposition analysis of embodied import and export emissions. The main findings are that the embodied import greenhouse gas emissions have increased by 37% whereas export emissions increased by only 3%, which is primarily driven by CO_2. The 2007 bilateral balances are positive with OECD countries but negative with economies such as Russia, Africa and China. The analyses demonstrate that the worsening of the ETB is to a large extent caused by the changing composition of trade: the Dutch economy increasingly exports clean products and imports dirty products.

Answer to research questions

Returning now to the main research question of this thesis, we will start by discussing the first sub question about the main causes for the existence of a gap between theory and practice.

First of all, as Chapter 3 demonstrated, there is a lot of confusion between theory and practice about *basic concepts* such as 'rent' and 'price'. For instance, a standard formulation according to the 2008 SNA and the SEEA CF is that the value of an asset consists of the sum of discounted resource rents. However, this use of the notion resource rent already seems to differ from how rent is usually understood in (environmental) economics where 'current rent' equates the current change in market value of the asset. We also saw in Chapter 3 that there are many different price notions: the SNA focuses on market prices for transactions defined as *the amounts of money that willing buyers pay to acquire something from willing sellers* (para 3.119). The SNA makes clear in the same paragraph, that this implies that market prices need not be similar to free market prices (assuming the existence of a competitive market), or the going price, or world market prices etc. Furthermore, in the SNA output is recorded at basic prices, while use of products is recorded at purchaser's prices. Differences between the two are due to taxes and subsidies, trade and transport margins, predictable quality changes (e.g. maturing wines), holding gains during storage etc. (ibid., para 3.148) On the other hand, many theoretical contributions require shadow prices; these will however be different from market prices due to distortions and externalities. Confusingly, shadow prices are sometime called 'accounting prices' (e.g. by Hamilton and Ruta, 2009; and Mäler et al., 2009). Quite often – as we saw in Chapter 3 – theoretical contributions have to resort to market prices by lack of observability of shadow prices. Finally, although not analyzed in detail in this thesis, the notion capital gain is also a frequent source for misunderstandings.

A second reason for the existence of the gap is due to the existence of *different contexts of use.* Chapter 3 distinguished – when discussing adjusting income measures for the cost of depletion – between assessments of i) social welfare

ii) sustainable income, and iii) aggregate economic activity.[1] Underneath these contexts of use lie different traditions which are concerned with different questions. A lot of the theoretical work on green accounting has clearly focused on measuring welfare, an analysis which requires a foundation in a macro economic model including welfare functions. Environmental accounting from its inception has been oriented towards estimating sustainable income departing from notions of Hicksian income, hence the initial focus on estimating a 'green GDP'. The SNA makes clear that it is about measuring economic activity and not about measuring welfare. Another example about the importance of distinguishing between different contexts of use was encountered in Chapter 4, when we discussed the valuation of ecosystem services. The literature on ecosystem services has often focused on what is referred to as total economic values which includes consumer surplus. This is understandable given that these values are often used for cost benefit analysis or environmental assessment strategies. On the other hand, the SNA (and SEEA) depart from the notion of market exchange value which excludes consumer surplus, and is often referred to as marginal valuation.[2]

A third reason for the gap may be due to *different objectives*: the focus of a statistical office is on publishing the best possible data for a single country. Furthermore, national accounts often focus on compiling estimates in volume terms, that is, excluding the change in value that is due to changes in the structure of prices. Estimating growth rates therefore often trumps estimating levels of macroeconomic indicators. It is only during a revision that national accounts are again benchmarked to underlying data sources. By contrast, the objective of theoretical contributions such as the World Bank's wealth accounts discussed in Chapter 5 (but also UNU-IHDP and UNEP, 2012) is on making a solid comparison across countries rather than across time for a single country. A key issue for the latter therefore becomes how to reconcile official data from individual countries with the need to consistently estimate data for a large panel of countries, relying on uniform and generic assumptions. The possibility to compare countries often trumps individual country estimates. As we have seen in Chapter 5, the discrepancies between official statistics and estimated data can be very significant, at least for the Netherlands. A similar example was encountered in Chapter 6 on input-output modeling. The use of MRIOs which combine and integrate multiple data sources from numerous countries often implies that significant adjustments are made to individual country

[1] Heal and Kriström (2005) distinguish between income as expenditure level that can be continued in the future and income as a welfare measure; Vellinga and Withagen (1996) distinguish between three purposes of NNP: welfare, cost-benefit analysis and sustainability.

[2] Confusingly, shadow prices are also marginal in the sense that they can be defined in terms of partial derivatives of the welfare function e.g. as the resulting welfare effect when a constraint is relaxed with one unit.

data, as evidenced by the example of the Netherlands, when comparing with the GTAP database.

The second sub question is whether it would be possible to strengthen environmental accounting practices by underpinning them with a theoretical foundation. The possibilities here seem limited.

First, as explained in Chapter 2, environmental accounting is essentially a satellite system to the SNA based upon the same principles. As indicated in Chapter 1 the SNA has gradually moved towards a system that aspires to be multipurpose and theory neutral. It derives its legitimacy not so much from an underlying theoretical framework but rather from the fact that it is the outcome of an intergovernmental process that takes comments by many different stakeholders into account. Moreover, it is not a fixed document but being revised every 10–15 years. A fortiori, a large part of the environmental accounts derives its legitimacy from being a satellite system to the SNA. Severing ties with the SNA, for instance by placing environmental accounting on a sustainable income footing, may risk undermining its legitimacy and appeal.

Second, an issue that is easily overlooked is that the SNA and SEEA cover a wide range of issues (choice of units; system boundaries; classifications; etc.) that is not so easily replaceable by a single theory. Moreover, the SNA and SEEA are no longer stand-alone frameworks; they are part of the whole edifice of economic statistics which includes manuals on balance of payments, government statistics, index theory etc. And, as the discussion on whether or not to capitalize R&D expenditure has shown in the context of the European System of Accounts, even though there could be compelling theoretical reasons, the availability of reliable country data is also of great concern.[3)]

Third, as exemplified by the discussions in Chapters 3 and 5 there are several rival theories (e.g. inclusive wealth; comprehensive wealth; sectoral income theory). Obviously, a difficulty for choosing a theoretical underpinning for accounting is that the accounts may become vulnerable to criticism of the underlying theory. The adoption of the SEEA CF as an international statistical standard bears testimony to the fact that environmental accounting can be successful without a strong theoretical underpinning.

3) Although in this case the decision was eventually taken in favor of capitalization.

We are now in a position to answer the main question of this thesis: what are the possibilities to narrow the gap between theory and practice in green/environmental accounting? Due to the existence of different contexts of use and objectives and difficulties in underpinning environmental accounting practices by a foundation from the theoretical literature, there are few direct possibilities. The reconciliation should be primarily sought in enhancing mutual recognition of different contexts and approaches. Practitioners and researchers often seem to not have been fully aware of the different contexts they operate within. As an illustration, one of the respondents to the Global Consultation held within the Netherlands on the draft SEEA Experimental Ecosystem Accounting guidelines expressed that he found the distinction between total and marginal values an 'eye opener'. This mutual recognition will be enhanced by increased cooperation between the statistical and research communities.

Policy implications

Turning now towards policy, several lessons can be learned from this thesis.

First of all, given the existence of multiple contexts of use and valuation principles, valuation exercises could be designed in such a way that the results are useful for multiple contexts of use. For instance, ecosystem services valuation studies could be designed in such a way that they are useful both for ecosystem assessments and for accounting. To give an example, suppose we want to value the recreational service of a specific forest that is open to visitors without any entree fee. An outcome in the form of a euro value per hectare based on a contingent valuation study may not be directly useable for accounting as it will include consumer surplus. However, a demand curve which specifies the expected number of visits as a function of hypothetical entree fees would be very useful. Indeed, given such a demand curve, taking the actual number of visits multiplied by the corresponding price from the demand curve would allow to obtain an estimate of the SNA consistent value of the amenity service.

Second, the thesis shows that the potential of a single 'green GDP' type of indicator is limited. As shown in Chapter 2, although quite a few countries have experimented with compiling 'green GDP' types of indicators, there are few (e.g. Mexico) examples of successful integration in policy making. There are a number of reasons that may help explain this. First of all, the definition of 'green GDP' itself is unclear. For instance, as shown in Chapter 3, the precise definition and recording of depletion remains (at least until the standardization reached by the SEEA CF) controversial. The recording of degradation and/or ecosystem services, which was discussed in Chapter 4, proves even more difficult, as we have seen that the type of adjustment critically depends on the chosen approach. Furthermore, there may also be various consistency issues at stake between the methods used for valuing degradation and the accounting framework.[4)] Furthermore, as the Stiglitz-Sen-Fitoussi report (Stiglitz et al., 2009) argued, even if we were to have perfect 'green GDP' measures, it would not tell us whether we are becoming more sustainable or not.[5)] Moreover, several studies (e.g. Brouwer et al., 1999; Advisory Committee, 2002) have concluded that adjusting GDP quickly requires modeling which may lie beyond the realm of statistical offices (in fact the 2003 SEEA referred to this as 'greened economy modeling'). Therefore, it remains to be seen to what extent estimating adjusted income aggregates should be the responsibility of the statistical community. Finally, when 'green GDP' is approached from a theoretical model (as shown in the Appendix of Chapter 5) it faces the issue described by Pezzey (2003, p.666) as "dependency of the adjustment prescription on model specifications"; the correction terms required to calculate a 'green GDP' depend on the choice of the underlying model.

Future outlook

Chapter 2 demonstrated that environmental accounting is a growing area of statistics. There appears to be increasing international recognition of the importance of environmental accounting as a framework for deriving indicators: indicators to measure societal progress as expressed by Stiglitz et al. (2009); sustainable development indicators (UNECE, 2009); and indicators for assessing green growth (OECD, 2011b).

In terms of the type of indicators, the focus has gradually shifted from 'green GDP' type of indicators towards wealth based indicators (UNU-IHDP and UNEP, 2012; World Bank, 2011); footprint indicators and efficiency/productivity types of indicators as evidenced by green growth strategies (OECD, 2011a). Moreover, the focus has also shifted from finding a single summary or headline indicator (such as

[4)] For instance, when degradation costs are estimated using impacts on human health, there are two issues. First, human capital as such is not recognized as an asset within the SNA so it is a bit cumbersome to degrade it. Second, part of the impacts on health may be already reflected in the accounts in the form of reduced output.

[5)] The report credits the World Bank's adjusted net savings indicator and the ecological footprint indicator for addressing this issue. However, as argued by Edens and De Haan (2010), we believe that the SSF report does not do full justice to the SEEA CF. As evidenced by the fact that SEEA is classified under 'adjusted GDPs', the report takes a narrow view of the types of indicators that the system has to offer. Essentially the system is reduced to what may have been its primary purpose in earlier versions –'green GDP' – but as we have illustrated in Chapter 2 does not reflect its current multipurpose character. The SSF report somehow misses one of the main building blocks of SEEA – the asset accounts - which can be used for wealth accounting.

'green GDP' or a composite index) towards choosing a number of headline indicators (e.g. recommendation 11 of the SSF report calls for a dashboard of indicators).

In order to fulfill these policy demands, this thesis has argued that it is of paramount importance to stimulate enhanced cooperation between theorists and practitioners in environmental and green accounting, in particular in the following domains.

First of all, the area of environmentally extended input-output analysis. There is a growing interest in consumption based indicators such as carbon footprints, virtual water or indirect resource use. As the calculation of these indicators requires environmental data that is integrated with economic statistics, there is a growing demand for environmental accounting data. The current state of the art is the use of Multi-Regional Input-Output (MRIO) tables which provide an integrated trade linked structure of the world economy (see Hoekstra et al., 2012). The development and maintenance of MRIOs lies due their huge data requirements typically beyond the scope of individual NSIs. Moreover, many individual IO tables have certain specifics (e.g. certain economic sectors are recorded gross instead of net) which may severely distort the outcomes of a global analysis when such details are overlooked. With the replacement of the 1993 SNA by the 2008 SNA guidelines, which strictly follow criteria of economic ownership, national accounts are increasingly disconnected from the underlying physical flows (Van Rossum et al., 2010), which will also pose additional challenges for the field of input-output analysis. Addressing those issues would benefit from a collaborative effort.

A second area of cooperation lies in wealth accounting. Several recent contributions (e.g. Dasgupta, 2009; Heal and Kriström; Arrow et al., 2003a) question the welfare economic theory of green accounting, and emphasize the importance of using wealth based measures. This approach holds the promise of uniting two traditions mentioned above, of assessing sustainability and welfare.[6] This seems to have lead to a surge of interest in so-called wealth accounting of late. An important aspect will be to gain a better understanding how wealth estimates compiled *for* countries are related to national accounts data on balance sheets compiled *by* countries – the third context of use that was distinguished. When the valuation principles and/or the asset boundary used by national accountants differ from those employed in research studies there is a risk that users may become confused.

[6] Dasgupta (2009) shows that wealth defined as the sum of stocks at shadow prices moves in the same direction as well-being, 'one is dual to the other' (ibid., p.5).

The lack of official statistics in the area of wealth accounting continues to be an issue that needs further scrutiny.

A third area of cooperation is ecosystem accounting. So far, the use of spatially explicit data such as remote sensing data within environmental accounting has been limited. This may change, as land and ecosystem accounting is an emerging area within environmental accounting. The use and integration of such data set however requires further cooperation between multiple scientific disciplines (Obst et al., 2013).

8. Nederlandse samenvatting

Verbinden van theorie en praktijk op het gebied van milieurekeningen

Elke economie is voor haar functioneren afhankelijk van de inzet van natuurlijk kapitaal. Niet alleen vanwege het gebruik van grondstoffen, maar ook voor de absorptie van afval en emissies. Dergelijke afhankelijkheden komen echter niet goed tot uitdrukking in het systeem van nationale rekeningen (System of National Accounts of SNA – UN et al., 2009) dat gericht is op markttransacties. Als een land bijvoorbeeld zou besluiten haar energie reserves versneld te winnen, dan zouden de volledige opbrengsten in het nationaal inkomen worden opgenomen, terwijl hier geen kostenpost tegenover staat. Dit is onbevredigend aangezien er op deze wijze een asymmetrie ontstaat tussen de behandeling van geproduceerd kapitaal (zoals een machine) waarop wel wordt afgeschreven, en natuurlijk kapitaal waarop niet wordt afgeschreven. Vanuit een duurzaamheidsperspectief bezien, zorgt het er bovendien voor dat als een dergelijke voorraadonttrekking de bestendigheid van het inkomen ondermijnt, macro-economische indicatoren misleidende signalen kunnen geven aan beleidsmakers.

Dergelijke vraagstukken worden al geruime tijd onderzocht door enerzijds statistici (met name de nationale- en milieurekenaars) en anderzijds (milieu-)economen. In dit proefschrift wordt een onderscheid gemaakt tussen *green accounting* (groene rekeningen – zoals dit terrein vaak omschreven wordt in de onderzoekswereld/theoretische literatuur) en *environmental accounting* (milieurekeningen – ook wel milieu-economische rekeningen – de notie die doorgaans gebruikt wordt in de statistische gemeenschap en de empirische literatuur). Terwijl beide werkvelden een gedeelde ambitie hebben om betere indicatoren te ontwikkelen om (materiële) welvaart en duurzaamheid te meten, bestaan er grote verschillen in gehanteerde uitgangspunten en methoden. De theoretische literatuur is van oudsher gericht op het bestuderen van de relatie tussen begrippen als welzijn, inkomen en vermogen, door gebruik te maken van theoretische modellen. Hierbij worden onderwerpen als de uitputting van natuurlijke hulpbronnen, vervuiling en de behandeling van ecosysteemdiensten bestudeerd (e.g. Dasgupta en Heal, 1974, Weitzman, 1976; Hamilton, 1996; Arrow et al., 2003a; Asheim en Wei, 2009; Dasgupta, 2009; Barbier, 2013). De statistische gemeenschap kiest vaak een meer pragmatische aanpak, gericht op de vraag hoe het gebruik van natuurlijk kapitaal het beste te integreren is in de nationale rekeningen.

Er bestaat echter, zoals opgemerkt door Heal en Kriström (2005, p.1151) een duidelijke kloof tussen theorie en praktijk. De milieurekeningen worden verweten onvoldoende duidelijk te maken wat ze precies beogen te meten, bij gebrek aan een theoretische onderbouwing. De theoretische literatuur gebruikt daarentegen dikwijls onrealistische veronderstellingen zoals een optimale werking van de economie. Voorts bestaat er verschil van inzicht of een theoretisch fundament überhaupt wenselijk is. Tegelijkertijd is er de laatste jaren een groeiende erkenning van het belang van milieurekeningen bijvoorbeeld binnen de 'GDP and Beyond Roadmap' (European Communities, 2009).

De voornaamste motivatie voor het schrijven van dit proefschrift is derhalve de wens deze kloof zo mogelijk te verkleinen. De onderzoeksvraag waar dit proefschrift zich mee bezig houdt luidt als volgt:

Welke mogelijkheden bestaan er om de kloof tussen theorie en praktijk op het terrein van de groene/milieurekeningen te overbruggen?

Met de tegenstelling tussen theorie en praktijk wordt in het proefschrift gedoeld op het bestaan van grotendeels gescheiden werelden (*grosso modo* de statistische wereld en de onderzoekswereld) die elk gekarakteriseerd kunnen worden door het gebruik van eigen conventies en principes, zoals die tot uiting komen in handboeken, voorschriften en, respectievelijk, de empirische en theoretische literatuur. Om de onderzoeksvraag te kunnen beantwoorden worden de volgende twee deelvragen onderscheiden:

Wat zijn de voornaamste oorzaken voor het bestaan van deze kloof tussen theorie en praktijk op het terrein van de groene/milieurekeningen?

Kunnen milieurekeningen praktijken verstevigd worden door ze te voorzien van een theoretisch fundament?

Aangezien het onderzoeksterrein van de milieurekeningen inmiddels vrij omvangrijk is, is deze onderzoeksvraag ingeperkt door het onderzoek met name te richten op waarderingsvraagstukken (ofwel monetariseren). Dit is ook mede ingegeven doordat op het terrein van de fysieke milieurekeningen inmiddels internationale standaardisatie is bereikt. De gehanteerde onderzoeksmethode bestaat eruit de onderzoeksvraag te onderzoeken in een viertal verschillende domeinen: het klassieke probleem van het waarderen van de uitputting (*depletion*) van grondstoffen; het nieuwe terrein ecosysteem rekeningen; de bepaling van het nationaal vermogen (*wealth accounting*); en, een toepassing van de milieurekeningen in de vorm van milieu input-output analyse.

Hoofdstuk 1 beschrijft de motivatie, beleidsrelevantie en onderzoeksvragen van dit proefschrift. Hoofdstuk 2 biedt verdere achtergronden en contextuele informatie. Het begint met een beknopt historisch overzicht over de ontwikkeling van milieurekeningen, gevolgd door een uitleg over het systeem van milieurekeningen. Milieurekeningen worden in dit proefschrift gedefinieerd als een zogenaamde satellietrekening van het SNA, zoals beschreven in het *System of Environmental-Economic Accounting* (SEEA – UN et al., 2012/2013). Satellietrekeningen respecteren de basisdefinities en classificaties die aan de nationale rekeningen ten grondslag liggen, maar staan tegelijkertijd enige mate van flexibiliteit toe. Bijvoorbeeld door het gebruik van een uitgebreid productiebegrip, een andere keuze van de grens van activa, of door het geven van aanvullende classificaties (Edens en De Haan, 2010). Daarnaast bevat hoofdstuk 2 een overzicht van de ontwikkeling van milieurekeningen programma's in landen, met een focus op twee aspecten die vanuit waarderingsgperspectief interessant zijn: ervaringen van landen met het bepalen van het nationaal vermogen (*wealth accounting)* en de ervaringen met het schatten van een groen BBP (*'green GDP'*).

Hoofdstuk 3 onderzoekt het klassieke probleem van de extractie van een niet-hernieuwbare hulpbron (vanwege het belang voor de Nederlandse economie, aardgas) en analyseert een viertal recente voorstellen die zijn gedaan om de kosten van uitputting te bepalen door ze concreet uit te rekenen aan de hand van data over de ontwikkeling van de Nederlandse aardgasreserves. De belangrijkste bevindingen zijn dat het corrigeren voor de kosten van uitputting zou leiden tot belangrijke aanpassingen van zowel het niveau als ook de nominale groei van het Nederlandse netto nationaal inkomen, met een sterke afhankelijkheid van de gekozen benadering.

Hoofdstuk 4 onderzoekt de relatie tussen theorie en praktijk op het opkomende terrein van ecosysteemrekeningen, waarin natuurlijk kapitaal niet langer reductionistisch beschreven wordt (een bos als verzameling hout, grond, water etc.) maar eerder als een organisch geheel (een bos als ecosysteem dat een bundel aan ecosysteemdiensten levert) (MA, 2005; TEEB 2010). Dergelijke ecosysteemdiensten zoals het vastleggen van koolstof, het leveren van water, het bieden van een mooie omgeving voor recreatie, zijn echter niet goed zichtbaar in de nationale rekeningen omdat ze doorgaans ongeprijsd zijn. In het hoofdstuk worden vier belangrijke methodologische problemen in het ontwikkelen van ecosysteemrekeningen geïdentificeerd en geanalyseerd: de definitie van ecosysteemdiensten, de toewijzing van ecosysteemdiensten aan institutionele sectoren, de behandeling van degradatie en rehabilitatie van ecosystemen; en het waarderen van ecosysteemdiensten in overeenstemming met SNA principes. Er worden voorstellen uitgewerkt hoe ecosysteemdiensten, degradatie en rehabilitatie in een rekeningenstelsel kunnen worden geïntegreerd. Nieuwe aspecten hierin zijn om als uitgangspunt te nemen dat de meeste ecosystemen sterk beïnvloed zijn door mensen en dat het leveren van ecosysteemdiensten daarom afhankelijk is van zowel natuurlijke processen als van het menselijke beheer van ecosystemen; en om de toekenning van ecosysteemdiensten aan economische sectoren te laten afhangen van het type en het karakter van de dienst.

Hoofdstuk 5 analyseert schattingen die de Wereldbank (2011) onlangs heeft gepubliceerd aangaande het nationaal vermogen en de gecorrigeerde netto-besparingen (*adjusted net savings*) voor meer dan 120 landen, door de schattingen voor Nederland zo goed mogelijk te vergelijken met relevante officiële Nederlandse statistieken. De belangrijkste empirische bevindingen zijn: i) de gecorrigeerde netto-besparingen bedragen ongeveer 13 procent van het bruto nationaal inkomen, in vergelijking met 15 procent volgens de Wereldbank, als gevolg van hogere degradatiekosten vanwege het opnemen van een breder scala aan luchtemissies; ii) het zogenaamde immateriële kapitaal (dit bestaat grotendeels uit menselijk kapitaal) vormt een veel kleiner deel van het nationaal vermogen (59 procent indien de methode van de Wereldbank wordt gevolgd; 66 procent bij het gebruik van directe ramingen) dan volgens de Wereldbank (ongeveer 80 procent). Deze discrepantie is grotendeels te verklaren door het gebruik van generieke aannames door de Wereldbank, alsmede een verschil in gebruikte bronnen. Hierdoor ontstaan er grote verschillen in de waardering van grond in stedelijke gebieden, en in geproduceerd en financieel kapitaal. Daarnaast worden de onderwijsuitgaven door de Wereldbank meegenomen in de bepaling van het totale vermogen, terwijl deze volgens het onderliggende model beschouwd zou moeten worden als een investering in menselijk kapitaal. Het hoofdstuk eindigt met een kritische bespreking van een artikel van Ferreira en Moro (2011) dat een vergelijkbare analyse maakt voor Ierland, door te beargumenteren dat de negatieve gecorrigeerde netto-besparingen die zij vinden veroorzaakt worden door – vergeleken met de literatuur – zeer hoge degradatiekosten.

Hoofdstuk 6 bevat een toepassing van milieurekeningen in de vorm van milieu input-output analyse. Met deze technieken kan berekend worden wat de wereldwijde uitstoot is van emissies ten behoeve van Nederlandse consumptie. Dit kan vervolgens worden uitgedrukt in de vorm van een *carbon footprint* of een *emissiehandelsbalans*. De laatste geeft het verschil aan tussen de emissies die een land exporteert – middels producten bestemd voor consumptie in andere landen – en vice versa importeert. Bilaterale emissiehandelssaldi voor Nederland met 17 landen/regio's worden opgesteld en vergeleken voor twee verschillende jaren, 1996 en 2007, en voor drie verschillende broeikasgassen. Door middel van verschillende technieken wordt inzicht verkrijgen in deze balansen en hoe ze veranderen door de tijd. De belangrijkste bevindingen zijn dat de uitstoot van de import emissies is gestegen met 37 procent, met name vanwege de toename van CO_2, terwijl de uitstoot van de export emissies steeg met slechts 3 procent. De bilaterale saldi van 2007 zijn positief met OESO-landen, maar negatief met economieën zoals Rusland, Afrika en China. De analyses tonen aan dat de verslechtering van de emissiehandelsbalans voor een groot deel veroorzaakt wordt door de veranderende samenstelling van de handel: de Nederlandse economie exporteert steeds meer schone producten en importeert steeds meer relatief vuile producten.

In Hoofdstuk 7 worden de onderzoeksvragen van dit proefschrift beantwoord. Er worden drie verklaringen gegeven voor het bestaan van een kloof tussen theorie en praktijk. Allereerst, zoals bleek uit de analyse in hoofdstuk 3, bestaat er veel verwarring tussen theorie en praktijk over elementaire begrippen als overwinst (*rent*) en prijs. Het SNA richt zich op marktprijzen voor transacties, terwijl veel theoretische bijdragen schaduwprijzen vereisen, die doorgaans zullen verschillen van marktprijzen als gevolg van marktverstoringen en het optreden van externe effecten.

Een tweede reden voor het vormt het bestaan van verschillende gebruikscontexten. Hoofdstuk 3 maakt duidelijk dat er wat betreft het meten van nationaal inkomen een onderscheid gemaakt moet worden tussen het meten van welvaart, duurzaam inkomen, of economische activiteit. Veel theoretische bijdragen hebben van oudsher een focus op het meten van welvaart, hetgeen het gebruik van een onderliggend macro-economisch model met daarin welvaartsfuncties vereist. Milieurekeningen zijn vanaf het begin georiënteerd geweest op het bepalen van een duurzaam inkomen – vandaar ook de aanvankelijke focus op ‘groen BBP’. Het SNA zelf maakt duidelijk dat het niet haar bedoeling is om welvaart te meten, maar dat het primair beoogt economische activiteit te meten. Een ander voorbeeld waaruit het belang blijkt van het onderscheid van gebruikscontexten, treffen we aan in hoofdstuk 4, waar de waardering van ecosysteemdiensten wordt besproken. De literatuur over ecosysteemdiensten is vaak gericht op wat wordt aangeduid als de totale economische waarde, die derhalve het consumentensurplus omvat. Dit is begrijpelijk, aangezien dergelijke waarderingsstudies vaak gebruikt worden voor het maken van een kosten-batenanalyse. Aan de andere kant zijn het SNA (en SEEA) gestoeld op een marginale waarderingsgrondslag, waarmee het consumentensurplus juist wordt uitgesloten.

Een derde reden is dat er verschillende doelstellingen zijn: de focus van een statistisch bureau ligt doorgaans op het publiceren van de best mogelijke data voor één land. Bovendien is het ramen van economische groeicijfers doorgaans belangrijker dan het ramen van het niveau. Alleen tijdens revisies worden de nationale rekeningen opnieuw geijkt aan de niveaus van de onderliggende gegevensbronnen. Daarentegen is de doelstelling van veel theoretische bijdragen (zoals de *wealth accounts* van de Wereldbank en UNEP, zoals besproken in hoofdstuk 5) juist het maken van een solide vergelijking tussen landen, in plaats van in de tijd voor één land. Er ontstaat dan ook spanning tussen het gebruik van officiële gegevens van afzonderlijke landen en de wens om gegevens voor een groot panel aan landen te schatten met een beroep op generieke veronderstellingen waardoor de schattingen vergelijkbaar zijn. Zoals we gezien hebben in hoofdstuk 5, kan de discrepantie tussen de officiële statistieken en dergelijke geschatte gegevens significant zijn, althans voor Nederland. Eenzelfde conclusie kan worden getrokken uit de discussie in hoofdstuk 6 over input-output analyse. Het gebruik van multi-regionale input-output tabellen, waarin meerdere gegevensbronnen uit verschillende landen worden gecombineerd en geïntegreerd heeft vaak tot gevolg dat significante aanpassingen worden gemaakt aan data van afzonderlijke landen, zoals bleek uit het voorbeeld van Nederland.

De tweede deelvraag, of het mogelijk is om milieurekeningen praktijken te verstevigen door ze te voorzien van een theoretisch fundament, kan als volgt beantwoord worden: deze mogelijkheden zijn beperkt.

In de eerste plaats komt dit zoals in hoofdstuk 2 is aangegeven doordat de milieurekeningen een satelliet vormen van het SNA. Zoals in hoofdstuk 1 is aangegeven, heeft het SNA zich geleidelijk ontwikkeld van een systeem afkomstig uit een Keynesiaanse traditie, naar een systeem dat juist streeft multifunctioneel en in zekere zin 'theorie-neutraal' te zijn. Het SNA ontleent haar legitimiteit niet zozeer aan een onderliggend theoretisch fundament, maar uit het feit dat het de uitkomst is van een intergouvernementeel proces waarin belangen van diverse partijen zijn gewogen. Bovendien is het SNA niet statisch, maar wordt het elke 10–15 jaar herzien, en bestaat er inmiddels ook een rijke traditie. A fortiori, een deel van de milieurekeningen ontleent haar legitimiteit weer aan het feit dat het een satellietsysteem is van het SNA. Indien de milieurekeningen gestoeld zouden worden op bijvoorbeeld een duurzaam inkomensbegrip dan zouden ze los komen te staan van het SNA en daarmee zou haar legitimiteit deels ondergraven worden.

Ten tweede, een kwestie die gemakkelijk over het hoofd gezien wordt is dat het SNA en SEEA een breed scala aan onderwerpen bestrijken (zoals de keuze van statistische eenheden; classificaties, etc.) waardoor ze ook niet zo eenvoudig te vervangen zijn door een enkele theorie. Bovendien is het SNA (en SEEA) niet langer een op zichzelf staand document, het is onderdeel geworden van een heel bouwwerk van economische statistieken (overheidsstatistieken, prijsstatistieken, statistieken over de betalingsbalans, etc.) De beschikbaarheid van betrouwbare gegevens is minstens zo belangrijk als theoretische overwegingen.

Ten derde, zoals is gebleken uit de discussies in hoofdstuk 3 en 5 bestaan er verschillende rivaliserende theorieën (zoals *inclusive wealth*; *comprehensive wealth*; *sectoral income theory*). Door het kiezen van een theoretische onderbouwing zouden de milieurekeningen ook kwetsbaar worden voor eventuele kritiek op de achterliggende economische theorie. Tot slot toont de erkenning van het SEEA Central Framework (UN et al, 2012) als internationale standaard ook aan dat milieurekeningen ook succesvol kunnen zijn zonder een sterk theoretisch fundament.

De centrale onderzoeksvraag van dit proefschrift kan derhalve als volgt worden beantwoord: vanwege het bestaan van verschillende gebruikscontexten en doelstellingen, alsmede moeilijkheden om het SEEA te stoelen op een theoretisch fundament, lijken er weinig directe mogelijkheden om de kloof tussen theorie en praktijk te dichten. Mogelijke overbrugging zal met name gezocht moeten worden in verbeterde en nauwere samenwerking tussen de statistische- en de onderzoeksgemeenschap.

Implicaties voor beleid

Er kunnen een aantal beleidsrelevante conclusies worden getrokken uit het onderzoek.

Allereerst, waarderingsstudies van ecosysteemdiensten zouden op een dusdanige manier ingericht en ontworpen kunnen worden dat de resultaten bruikbaar zijn voor meerdere gebruikscontexten, bijvoorbeeld voor zowel kosten-baten analyses als voor milieurekeningen.

Ten tweede toont het proefschrift aan dat het potentieel van een alomvattende 'groen BBP' indicator beperkt is. Uit het overzicht in hoofdstuk 2 blijkt dat, alhoewel nogal wat landen ervaring hebben opgedaan met de compilatie van dergelijke indicatoren, er weinig voorbeelden zijn (wellicht alleen Mexico) van succesvolle integratie van een dergelijke indicator in beleid. Hier zijn een aantal mogelijke redenen voor. In de eerste plaats is de definitie van 'groen BBP ' niet altijd duidelijk. Zoals is gebleken uit hoofdstuk 3, blijkt de precieze definitie van uitputting al controversieel (althans tot de standaardisatie hiervan in SEEA CF), laat staan de mogelijke opname van de ecosysteemdiensten of degradatie van ecosystemen, zoals besproken in hoofdstuk 4. Voorts, zoals het Stiglitz-Sen-Fitoussi rapport beargumenteert (Stiglitz et al.. 2009), zelfs als we in staat zouden zijn om ons 'groen BBP' perfect te schatten, dan nog zou het ons niet kunnen vertellen of de economie duurzamer wordt of niet. Verschillende studies (Brouwer et al., 1999;

Advisory Committee, 2002) hebben geconcludeerd dat het schatten van een 'groen BBP' het bestaan van een model vereist, en dat een dergelijke exercitie daarom eigenlijk buiten het domein van de statistiek ligt (het 2003 SEEA spreekt dan ook over *greened economy modelling*). Tot slot speelt het probleem, dat als het 'groen BBP' wordt benaderd vanuit een theoretisch model (zoals bijvoorbeeld beschreven in de appendix van hoofdstuk 5) de definitie van 'groen BBP' afhankelijk wordt van de gebruikte modelspecificaties (zie Pezzey, in Perman et al., 2003, p.666).

Vooruitzicht

Hoofdstuk 2 heeft aangetoond dat de milieurekeningen een statistisch terrein is dat zich mag verheugen in groeiende belangstelling. Ook is er een toenemende internationale erkenning van het belang van milieurekeningen als een raamwerk waaruit diverse indicatoren kunnen worden afgeleid: indicatoren voor het monitoren van maatschappelijke vooruitgang (Stiglitz et al., 2009); duurzame ontwikkeling (UNECE, 2009), en groene groei indicatoren (OECD, 2011b). Om aan deze beleidseisen te kunnen voldoen, betoogt dit proefschrift dat het belangrijk is om een nauwere samenwerking te bewerkstelligen tussen statistici en onderzoekers, waarbij gedacht kan worden aan de volgende deelterreinen.

Allereerst op het gebied van milieu input-output analyse. Er is een groeiende belangstelling voor indicatoren die milieuvervuiling relateren aan consumptie in plaats van productie, zoals carbon footprints, virtueel water, of ramingen van indirect materiaal gebruik. Dergelijke berekeningen vereisen het bestaan van milieugegevens die zijn geïntegreerd met economische statistieken, en leiden derhalve tot een groeiende vraag naar data afkomstig uit milieurekeningen. De huidige *state of the art* is het gebruik van multi-regionale input-output (MRIO) tabellen (zie Hoekstra et al., 2012). De ontwikkeling en het onderhoud van MRIOs ligt echter vanwege hun enorme data vereisten meestal buiten het bereik van individuele statistische bureaus. Daarnaast speelt het probleem dat, ten gevolge van de nieuwe 2008 SNA richtlijnen die strikt het criterium van economische eigendom volgen, de nationale rekeningen steeds meer losgekoppeld raken van de fysieke stromen naar een land zoals beschreven in statistieken van internationale handel (Van Rossum et al.. 2010). Dit zal in de nabije toekomst leiden tot diverse uitdagingen op het gebied van input-output analyse, die sterk gebaat zouden zijn bij een gezamenlijke aanpak.

Een tweede gebied van samenwerking ligt op het terrein van het bepalen van het nationale vermogen. Verschillende recente bijdragen (bijvoorbeeld Dasgupta, 2009; Heal en Kriström; Arrow et al., 2003a) bekritiseren de welzijnseconomische benaderingen van *green accounting* en benadrukken het belang van een kapitaalgerichte benadering. Deze kapitaalgerichte benadering zou het wellicht mogelijk maken om de in hoofdstuk 3 onderscheiden tradities van enerzijds het meten van duurzaam inkomen en anderzijds welvaart te verenigen, en lijkt deels te hebben geleid tot de groeiende interesse in *wealth accounting*.[1] Een belangrijk aspect hierbij is om inzicht te krijgen in mogelijke verschillen tussen kapitaalramingen opgesteld *voor* landen en schattingen gemaakt *door* landen, bijvoorbeeld in de balansen van hun nationale rekeningen – de derde gebruikscontext die werd onderscheiden. Wanneer verschillende waarderingsprincipes en/of een verschillende grens van activa wordt gebruikt bestaat het risico dat gebruikers in verwarring raken. Het ontbreken van officiële statistieken op het gebied van diverse soorten van kapitaal voor een groot aantal landen blijft een probleem. Hiernaar zou, in het licht van de huidige interesse in *wealth accounting*, maar ook in het meten van productiviteit, meer onderzoek gedaan moeten worden.

Een derde gebied van mogelijke samenwerking ligt op het terrein van ecosysteem rekeningen. Tot dusverre is het gebruik van ruimtelijk expliciete gegevens zoals satelliet data (*remote sensing*) binnen de milieurekeningen beperkt. Met de groeiende interesse in ecosysteem rekeningen zou dit snel kunnen veranderen. Het gebruik en de integratie van dergelijke datasets vereist echter verdere samenwerking tussen verschillende wetenschappelijke disciplines (Obst et al., 2013).

[1] Dasgupta (2009) toont aan dat het nationaal vermogen, wanneer dit wordt gedefinieerd als de som van de kapitaalvoorraden gewaardeerd tegen schaduwprijzen, in dezelfde richting beweegt als welzijn ('dualiteit').

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Acronyms

ANS	Adjusted Net Savings
ETB	Emission Trade Balance
EEBT	Emissions Embodied in Bilateral Trade
GDP	Gross Domestic Product
GTAP	Global Trade Analysis Project
MRIO	Multi-Regional Input-Output model
NDP	Net Domestic Product
NNI	Net National Income
NPV	Net Present Value
NSI	National Statistical Institute
SEEA	System of Environmental-Economic Accounting
SEEA CF	System of Environmental-Economic Accounting Central Framework
SEEA EEA	System of Environmental-Economic Accounting Experimental Ecosystem Accounting
SNA	System of National Accounts
WAVES	Wealth Accounting and the Valuation of Ecosystem Services
TEEB	The Economics of Ecosystems and Biodiversity

Acknowledgements

For a long time, I've felt as an academic in search of a suitable research subject. Although to the regret of my former graduation thesis supervisor Jos Uffink, I decided not to continue research in the foundations of physics, I feel much indebted for his relentless dedication to science and his high standards of academic scholarship. These are key values that still inspire me in my professional life to date. Gradually, I became more and more interested in the field of economics, which was enhanced during my work for the CPB Netherlands Bureau for Policy Analysis. My search ended, when I was introduced to the topic of environmental accounting after joining the United Nations Statistics Division (UNSD) in New York in 2006. I sincerely thank Ivo Havinga and Alessandra Alfieri of UNSD for giving me this opportunity.

The reason I like the topic of environmental accounting so much is not just the fact that it brings together a range of scientific disciplines which interest me: economics; statistics; environmental science; but perhaps most importantly, the type of reasoning required is very philosophical. One of the first projects I worked on in UNSD was setting up a searchable archive with publications on environmental accounting. Searching for, reading, and eventually classifying about 300+ publications in environmental accounting proved to be a great foundation for my later research in this area. Alessandra was not only a great mentor, who taught me the basics of environmental accounts (as well as national accounts and classifications), she also has a tremendous knowledge about its history and development. I remember in particular that in preparation for one of the UNCEEA Bureau teleconferences we wrote a strategy paper about the need for a theoretical underpinning of SEEA. The response we got from the meeting was that this was *not* the way forward. In retrospect, that is the moment when the seeds of this thesis were sown.

We decided to move back to the Netherlands for a number of reasons. The opportunity to work at Statistics Netherlands in the environmental accounts team was one of the great attractors. Having worked on the conceptual side of environmental accounting in support of the SEEA revision for several years by then, I was thrilled to be able to finally touch real data and actually compile the accounts myself. I am much indebted to Peter van de Ven, the former Director of National Accounts, Anne Boelens, my team manager, and Gerard Eding, the current Director of National Accounts, for giving me this opportunity.

While still in New York, I had already contacted Cees Withagen asking him for advice regarding my initial idea to write a PhD thesis trying to bridge theory and practice in environmental accounting. In response to my question whether he could recommend someone in the Netherlands for supervising such a thesis, he immediately indicated that he would be interested himself. And I'm glad he did. Cees, as I look back, I have very pleasant memories of all the discussions we had together. I really learned a lot from your analytical ability to break down problems with pen and paper not distracted by words or graphs, focusing on the underlying mathematics. Your quick response time and feedback on manuscripts was also very stimulating. I would also like to express my gratitude to the Faculty of Economics and Business administration of VU University, in particular Piet Rietveld, for allowing me to become a PhD candidate.

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Writing a dissertation runs in the family: my grandfather Jacob Dirk Edens obtained his doctorate in 1938 at Groningen University in Medicine; my father-in-law Cees de Langen obtained his doctorate in 1979 at the VU University in Medicine. Without the unwavering support of my parents Roelf Edens and Annie Edens-Talsma this thesis would not have been possible.

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List of publications

Chapter 2

Edens, B., 2012b. National environmental accounting. In: Daniel Fogel, Sarah Fredericks, Lisa Butler Harrington, & Ian Spellerberg (Eds.), The Berkshire Encyclopedia of Sustainability: Vol. 6. Measurements, Indicators, and Research Methods for Sustainability. Great Barrington, MA: Berkshire Publishing.

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Chapter 7

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Curriculum Vitae

Bram Edens was born on August 8th 1976 in Harlingen, the Netherlands. After completing secondary education (VWO) at the Corderius College in Amersfoort, he studied at Utrecht University where he completed a master's degree in physics (with distinction), a master's degree in philosophy (cum laude) and a master's degree in philosophy of the exact sciences. His graduation thesis in the foundations of statistical physics – in particular studying the works of Ilya Prigogine – was awarded the best annual master's thesis in philosophy in the year 2001. He was elected to the University Council in 1999 and in 2000 appointed as a member of the Advisory Committee on the Quality of Education. Notwithstanding being awarded several scholarships – a London School of Economics Graduate Merit Award; a Nuffic Talent-program scholarship – he chose to enter the Japan Prizewinners Programme, a post graduate course in Japanese business and society. In Japan, he did research on Japanese independent politicians at the Matsushita Institute for Government and Management in Chigasaki, Japan.

Upon return to the Netherlands, in 2002 he started working at the CPB Netherlands Bureau for Economic Policy Analysis at first in the cost benefit analysis unit on developing an economic framework for the analysis of safety within long-term socio-economic scenarios, later in the applied general equilibrium modelling unit, where he modelled the impact of demographic uncertainty on social welfare systems. After marrying Maaike de Langen who was about to join the United Nations Development Program in Chad, he quit his job, drove through the Sahara to N'Djamena, where he ended up working as a financial controller for Doctors Without Borders in support of the refugees from the Darfur crisis. The couple relocated to New York City, when their first daughter Lude was expected, later followed by Amarins.

In 2006 he started working at the United Nations Statistics Division as an associate statistician in the area of environmental accounting, a topic that has fascinated and inspired him ever since. His responsibilities ranged from organizing meetings, writing research papers, training delegations, towards the construction of a knowledge platform. In 2009 after six years abroad the family moved back to the Netherlands, where he started working at Statistics Netherlands in the National Accounts Department as a statistical researcher and project coordinator in the area of environmental accounts. At the same time, he commenced working on his PhD thesis under supervision of prof. dr. Cees Withagen at the Faculty of Economics and Business Administration of VU University, Amsterdam. His current responsibilities include quality assurance of several environmental accounting topics and overseeing the work on the trade sector and the non-observed economy in the national accounts. He has consulted for the World Bank and the United Nations. He participated in the Editorial Board of the SEEA Experimental Ecosystem Accounting and, currently, is a member of the Policy and Technical Experts Committee of the World Bank's WAVES project. He has been a referee for several journals amongst others: Nature Climate Change; Environmental Science & Technology; Review of Income and Wealth; and, the Journal of Industrial Ecology.