The potential use of the Materials and Energy Flow Analysis (MEFA) framework to evaluate the environmental costs of agricultural production systems and possible applications to aquaculture

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ABSTRACT

Global aquaculture production is roughly doubling every ten years, thus raising sustainability concerns and motivating the development of tools to evaluate its environmental costs. This paper reviews the potential contribution of material flow analysis (MFA) and the human appropriation of net primary production (HANPP) in this context. MFA and HANPP are indicators included in the broad framework of material and energy flow analysis, abbreviated MEFA framework. MFA reports physical flows in tonnes per year through various socio-economic systems, including companies, economics sectors, households, national economies, villages or world regions. MFA is increasingly used to quantify material requirements and wastes/emissions of production systems, and can be used in comparative studies, given appropriate standardization. HANPP is an indicator of land-use intensity that is often used with reference to a defined territory. HANPP is the difference between the net primary productivity (NPP) of potential natural vegetation and the proportion of the NPP of actual vegetation remaining in the ecosystem after harvest. We conclude that the combined use of MFA and HANPP could support the comparative assessment of environmental costs of aquaculture, which would require further methodological developments.

INTRODUCTION

Globally, aquaculture supplies increasing amounts of aquatic animals such as fish, crustaceans and molluscs. More than 220 aquatic species are farmed, and the output

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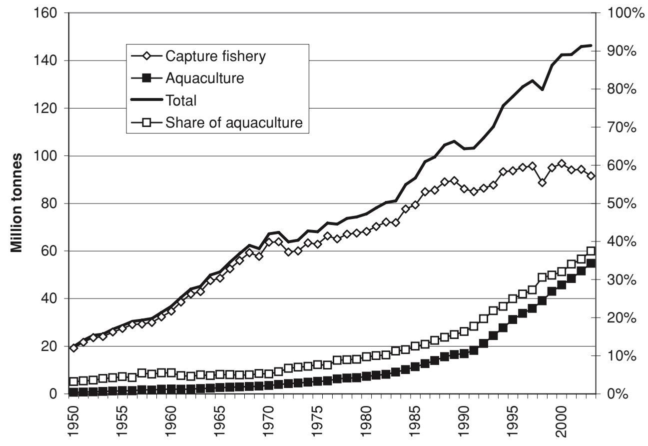
of aquaculture doubles roughly every 10 years (Naylor *et al.,* 2000), thus supplying valuable protein for human nutrition and economic benefits. Aquaculture currently accounts for more than one third of total global food fish production, and this share is rising constantly, as capture fisheries are stagnating due to the depletion of many fish stocks (Figure 1; Pauly *et al.,* 2002; Troell *et al.*, 2004).[2] Aquaculture production is forecast to continue to grow, with some scenarios assuming a total output of aquaculture in 2020 of over 80 Mt/yr (Delgado *et al.,* 2003; FAO, 2004).

The surging output of aquaculture systems has triggered concerns about environmental issues, such as pollution resulting from effluent discharge, loss of valuable habitats (e.g., mangrove forests), escape of farmed organisms affecting wild-living stocks ("biological pollution"), depletion of wild-living stocks due to the use of wild-caught juveniles in aquaculture systems, and environmental costs associated to feed procurement (Delgado *et al.,* 2003; Naylor *et al.,* 2000; Valiela , Bowen and York, 2001).

Many people hope that aquaculture can compensate shortfalls in ocean fish catches caused by deterioration of fish stocks (Delgado *et al.,* 2003; FAO, 2004). Aquaculture systems, however, often require feed containing fish meal derived from capture fisheries, so it very much depends on the origin of feed whether aquaculture can relieve pressures on wild fish populations. Fish meal derived from ocean fisheries is also used in some terrestrial animal rearing systems, above all for poultry, but some aquaculture systems currently require considerably more fish protein inputs than these terrestrial systems. Sometimes aquaculture systems, above those in which predatory species are cultivated, use about 5 times more protein from wild catch than their product contains (Naylor *et al.,* 2000; Pauly *et al.,* 2002).

All these issues raise concerns about the sustainability of aquaculture, thus motivating efforts to develop tools to evaluate its environmental costs. This paper reviews the potential value of using methods of material and energy flow accounting (MEFA) in this context. It should be clear, in any case, that these methods cannot address all the environmental issues associated to aquaculture, i.e. they have to be seen as complementary to other methods and tools.

FIGURE 1
Global production of capture fisheries and aquaculture 1950-2003. Data on the share of aquaculture (open squares) are given in percent of the total of capture fishery and aquaculture and refer to the secondary axis. Data source: downloaded from FAO's FIGIS homepage (http://www.fao.org/figis/servlet/static?dom=root&xml=index.xml) on 5 April 2006



[2] There are allegations of over-reporting by a major country that may affect figures reported in Figure 1 (Paul et al. 2002). Readers are advised to consult the scientific literature before using these data in cases where accuracy is crictical.

A REVIEW OF MEFA METHODS

As researchers increasingly acknowledge the problems associated with a "weak sustainability" perspective, above all the difficulties in adequately monetizing the value of ecosystem services and the questionable substitutability of human-made and natural capital, there is a rising demand for integrated (i.e. social-monetary-biophysical) analyses of socio-ecological systems (Martinez-Alier, 1999). Methods of "integrated environmental-economic accounting" are therefore increasingly used to analyse the interplay between economic activities and the environment. The "MEFA framework", an integrated toolkit to account for physical flows associated to socio-economic activities, plays an important role in this context (Haberl *et al.,* 2004b).

MEFA stands for "material and energy flow accounting," and it is based on the notion of socio-economic metabolism (e.g., Ayres and Simonis, 1994; Fischer-Kowalski, 1998; Fischer-Kowalski and Hüttler, 1998; Matthews *et al.,* 2000). The MEFA framework analyses important aspects of society-nature interaction by tracing socio-economic materials and energy flows and by assessing changes in relevant patterns and processes in ecosystems related to these flows (Haberl *et al.,* 2001b). It thus contributes to analyses of socio-economic activities from a "strong sustainability" perspective (Munasinghe and McNeely, 1995). Current work in this field seeks to analyse biophysical aspects of society in a way that is compatible with established tools for societal self-observation, above all, social and economic statistics upon which practically all modelling in economics and the social sciences rests. Such approaches were pioneered in the 1970's (Boulding, 1973; Ayres and Kneese, 1969).

Obviously, material and energy flows related to economic activities, although indispensable to "reintegrate the natural sciences with economics" (Hall *et al.* 2001), do not encompass society-nature interactions in their entirety. One important aspect that can not adequately be grasped by the socio-economic metabolism approach is land use – one of the most important socio-economic driving forces of Global Change (Meyer and Turner, 1994; Vitousek, 1992). Land use can be included in the MEFA framework by comparing ecosystem patterns and processes that would be expected without human intervention with those observable today. An example for this approach is the calculation of the "human appropriation of net primary production," or HANPP (Vitousek *et al.,* 1986).

The notion of a "MEFA framework" refers to an integrated, consistent accounting framework comprising data on socio-ecological metabolism. The MEFA framework is work in progress. Three parts of the framework have been proposed in considerable detail: (1) Material flow accounting, or MFA, has received most attention (e.g., Eurostat, 2001; Weisz *et al.,* 2005a). (2) Energy flow accounting (EFA) methods consistent with MFA have been proposed and applied (Haberl, 2001a; Haberl, 2001b; Haberl, 2006). (3) The Human Appropriation of Net Primary Production, or HANPP, proposed about 15 years ago (Vitousek *et al.,* 1986), has been further developed in a way that makes it consistent with material and energy flow accounting (Haberl *et al.,* 2001b). The MEFA framework is not necessarily complete with these three concepts. Expressing socio-economic metabolism not in terms of materials, but as carbon flow, would increase its usefulness for important applications, as would other, yet undeveloped accounting tools.

Material and energy flow analysis

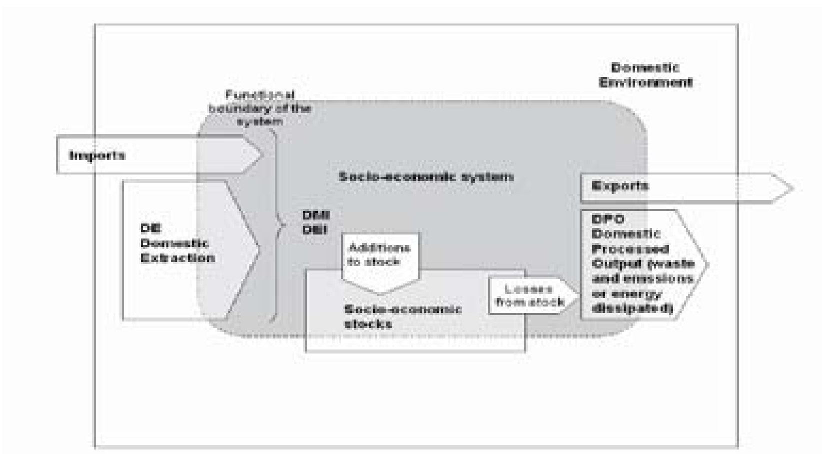
The general purpose of material flow accounting (MFA) is to quantify material inputs and outputs of socio-economic systems. MFA is a physical environmental accounting approach that tracks the use of materials by socio-economy systems from extraction to manufacturing, to final uses and disposal of emissions and wastes. It reports flows in physical units, usually metric tonnes per year, and can conceptually be linked to economic accounting frameworks, e.g. the System of National Accounts (SNA). The application of the mass balance principle ensures the consistency of the accounts. MFA can be applied to various scales and types of systems, e.g. companies, economics sectors, households, national economies, the world economy, or villages, cities, nation states and world regions.

MFA may include different types of materials. Coverage ranges from specific chemical elements or substances, for example copper (Graedel, 2002; Graedel *et al.*, 2002) or chlorine (Ayres, 1997a; 1997b), to all material inputs, including water and air, as in the case of the physical input-output table published by the German Federal Statistical Office (Stahmer, Kuhn and Braun, 1998).

Economy-wide material flow accounting – the application of MFA to national economies – is the most advanced type of MFA in terms of methodological harmonization and implementation into official statistics (Eurostat, 2001). Economy-wide MFA covers all material inputs (raw materials and imports), outputs (emissions and wastes, dissipative uses and exports) and net changes in socio-economic materials stocks, except for water and air. Hence, national MFA focuses on flows between a national economy and its environment which comprises both the natural environment and other socio-economic systems (Figure 2). National MFA usually does not include internal flows (i.e., flows within the economy, for example between economic sectors or actors), and flows within ecosystems on the national territory are outside its system boundaries and therefore not considered. Energy flow analysis (EFA) is a complementary tool used to account for the energy throughput of socio-economic systems. It uses the same definitions of system boundaries as economy-wide MFA, but is based on energy content (gross calorific value) of all flows as common currency (Haberl, 2001a, Haberl, 2001b).

National MFA methods date back to the 1960s (Ayres and Kneese, 1969; Gofman *et al.,* 1974; Wolman, 1965). The first national material flow accounts in the contemporary sense were published in the early 1990s for Austria, Germany, and Japan (Steurer, 1992; Japan Environment Agency, 1992; Bringezu, 1993; Fischer-Kowalski *et al.,* 1994). In the late 1990s the World Resources Institute coordinated the first comparative national material flow studies which included the United States of America, The Netherlands, Japan, Germany and Austria (Adriaanse *et al.,* 1997; Matthews *et al.,* 2000). A growing number of countries within the EU and the OECD implemented material flow accounting into their official environmental accounting program (see Weisz *et al.*, 2005a for a recent overview). In addition, MFAs have been published for a number

FIGURE 2
Material flow accounting scheme as used in national MFA



Source: simplified after Eurostat 2001 of developing countries, including Chile (Giljum, 2004); Brazil (Machado, 2001); Venezuela (Castellano, 2000); Philippines (Rapera, 2004); Thailand (Weisz, Krausmann and Sangkaman, 2006); and Laos (Schandl *et al.,* 2006). Eurostat has published economy-wide MFAs for all EU-15 member states in time series (Eurostat, 2002; Weisz *et al.,* 2005b), an extension to the ten new member states is in preparation. In parallel, the OECD is working on MFA databases for all OECD countries.

The publication of a methodological MFA guide by Eurostat (Eurostat, 2001) marks a major step forward in methodological harmonization of national MFA. Up to now this guide is the main methodological reference for the compilation of any economy-wide MFA. The Eurostat guide specifies the basic framework, its relation to the system of national accounts, defines the system boundaries to be applied (Figure 2), clarifies terminology, and suggests a number of aggregated indicators which can be derived from national MFA. The most decisive conceptual element of MFA is the definition of the system, because the system definition affects not only the results, but also predetermines potential uses of the data. The following features of national MFA systems have been identified as crucial: (1) compatibility of the accounts across countries and across time; (2) compatibility to the system of national accounts; (3) data availability and data quality; and (4) internal consistency of the framework. To achieve these goals the Eurostat guide on economy-wide material flow accounting proposes the following definition:

"The system boundary is defined:

1. By the extraction of primary (i.e., raw, crude or virgin) materials from the national environment and the discharge of materials to the national environment;
2. by the political (administrative) borders that determine material flows to and from the rest of the world (imports and exports). Natural flows into and out of geographical territory are excluded" (Eurostat, 2001, p 17).

The formulation of an exact definition of a crude or raw material is far from trivial, though. Statisticians and scientists have devoted a substantial amount of time to this question. Eventually it was concluded that a practical case-by-case definition meets the identified requirements best (for details see Ayres, Ayres and Warr, 2004; Fischer-Kowalski, 1998; Weisz *et al.,* 2005a). Eurostat, (2001) therefore proposes a number of practical conventions. Regarding agricultural systems these are: Agricultural plants are considered part of the natural system, therefore agricultural harvest as reported in agricultural statistics is accounted for as input from the natural system, while flows of nutrients between the soil and roots of agricultural plants are considered natural flows and are not part of MFA. Livestock is considered part of the economic system as long as its reproduction is under substantial human control. Consequently, uptake of grass by livestock from pastures and meadows has to be accounted for as a material input, whereas the production of meat and milk are internal flows of the economic system. Fish catch and hunted animals are considered as inputs into the system. All raw materials are conventionally accounted for in fresh weight, with the exception of grass harvest, fodder directly taken up by ruminants, and timber harvest. These latter raw materials are accounted for at a standardized water content of 15 percent (Eurostat, 2001; 2002).

At present, amendments and extensions of the original Eurostat guide regarding practical implementation of MFA including data sources as well as applicability to OECD countries are being developed by both Eurostat and the OECD in close cooperation. Eurostat installed an MFA task force consisting of representatives from national statistical offices and experts in material flow accounting, to discuss and solve open methodological questions. So far the task force met twice, in November 2004 and in January 2006, a third meeting is planned for autumn 2006. One issue that was raised in the meetings is the growing importance of aquaculture for the production of fish. It was concluded that fish from capture fishery (both sea and inland waters) is regarded as input to the system, whereas fish production form aquaculture is regarded as internal flow and is therefore not counted as input. The assumption is that aquaculture implies the provision of food and other inputs to the systems which are already counted for in other MFA sub-accounts. Therefore, adding up the produced fish and the necessary feed inputs would result in double counting.

One major use of national MFA, so far, has been the analysis of the economy in physical terms, and the creation of highly aggregated indicators for material use and material efficiency. Among the manifold results generated by this body of work we here stress only a few which are particularly relevant for agricultural productions systems. In pre-industrial economies, biomass is the main raw material used in providing goods and energy. The transition to an industrialized mode of production additionally requires large amounts of fossil fuels, construction minerals, metals and industrial minerals (Schandl and Schulz, 2002). This agro-industrial transition normally does not result in a reduction in the overall demand for biomass, but rather supports a shift in the demand patterns of biomass from technical energy to meat production (Krausmann, 2004). Overall, we see a constantly high contribution of biomass to the overall material and energetic metabolism of industrial economies. Since 1970 biomass contributed continuously about 25 percent to the domestic material consumption and the domestic energy consumption in the EU-15 (Weisz *et al.,* 2005a; Haberl *et al.,* 2006b). Animal fodder constitutes a growing share of agricultural biomass inputs. Trade volumes are increasing for agricultural products (as for almost all other materials as well). In the EU, biomass production still presents the most important single cause of competitive land occupation (see Weisz *et al.,* 2005a).

It is one of the conceptual strengths of the MEFA framework that it provides an overall picture of the physical economy in a way that is comparable across time and across countries. With regard to MFA in particular, the potential uses of this framework has been recognized recently by many countries as well as national and international organizations (e.g. the UN, OECD, EEA, US EPA, the G8), thus fostering programs aiming to implement MFA into official statistics in order to facilitate its utility for policy making. Among the policy uses of MFA, environmental issues are but one which are currently considered. Others are resource scarcity, evaluation of trade-offs between various policies, land management, substitution potentials or more generally providing new ways to think about the supply and demand of materials of our societies (National Academy of Sciences and National Research Council 2003, White House meeting on MFA, 2004; OECD, 2004; CEC, 2005). For example, in 2003 the Japanese government enacted 'the Basic Law for Establishing a Sound Material-Cycle Society' (OECD, 2004). The Japanese government set three quantitative sustainability targets for the period 2000 – 2010 and focused on the management of material flows.

However, to be of full use for such a broad spectrum of applications, the MEFA framework must be further developed. A number of possible directions are currently discussed: One is the attempt to provide a much higher resolution in terms of materials (Weisz *et al.,* 2005b). This implies the development of a standardized classification scheme for materials a pursuit already under way at Eurostat. Another important line of research is the development of methods to consistently account for the amount of raw material extraction that was needed to produce imported and exported goods, a goal that requires efforts to harmonize definitions of system boundaries (i.e. the stage in the socio-economic production process where the materials are extracted from the environment) as well as solutions to conceptual (e.g., treatment of byproducts, avoidance of double-counting) and data problems. This implies a combination of MFA and LCA methods, probably by making use of input-output analysis. We will return to this issue below.

The attempt to provide a picture of the whole economy implies, however, that flows which are small compared to total economic flows are hardly visible in national MFA.

Fish catch (excluding aquaculture), for example, amounts to only about 1percent of the total quantity biomass extracted in the EU-15, while 15 percent is timber, 49 percent crops, and 35 percent agricultural byproducts and grass. Increasingly, MFA is also used to quantify the material requirement as well as waste and emission generation of specific production systems. With such information, environmental pressures associated with the material and energy uses of production systems can in principle be identified, and – given appropriate methodological standardization – compared between different production systems. If the environmental costs of a specific production system, as in this case aquaculture, are the main focus, additional methodological adoptions are necessary (see Section 3).

The human appropriation of net primary production (HANPP)

In using the land, humans alter the production ecology of ecosystems in two interrelated ways: (1) by changing the productivity (NPP per unit area) of ecosystems and (2) by harvesting parts of the NPP. Both processes result in an alteration of the amount of NPP available in ecosystems as compared to their original status. The human appropriation of net primary production (HANPP) is an indicator for land-use intensity based on the measurement of changes in the availability of trophic (biomass) energy in terrestrial ecosystems induced through land-use induced changes in productivity and harvest. Technically, HANPP has been defined as the difference between the NPP of potential natural vegetation and the part of the NPP of the actually prevailing vegetation remaining in ecosystems (Figure 3, Haberl, 1997; Haberl *et al.,* 2004a) according to the following formulae:

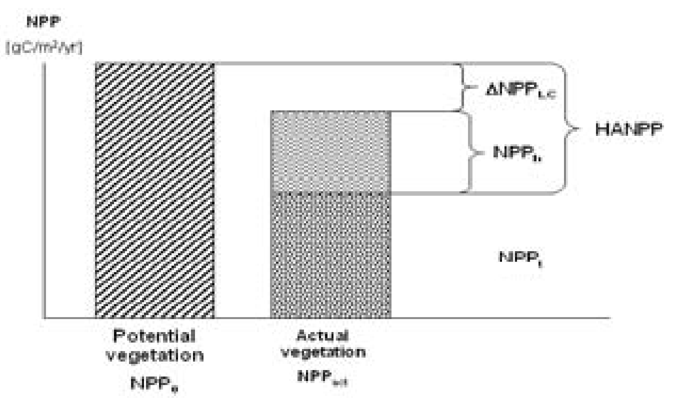
$$HANPP = NPP_0 - NPP_t \quad \text{with} \quad NPP_t = NPP_{act} - NPP_h$$

in which NPP_0 denotes the NPP of potential natural vegetation, NPP_t the NPP remaining in ecosystems, NPP_{act} the NPP of the currently prevailing vegetation and NPP_h the amount of NPP harvested by humans. If we denote as ΔNPP_{LC} the changes in productivity induced by land use ($=NPP_0 - NPP_{act}$) we get the following formula:

$$HANPP = \Delta NPP_{LC} + NPP_h$$

HANPP may be expressed as an absolute amount of dry matter biomass (kg dry matter), carbon contained in biomass (kgC), energy equivalent of biomass (J) or as a percentage of NPP_0. HANPP can be assessed for any defined area of land and can

FIGURE 3
Definition of HANPP as the difference between potential NPP and NPP actually remaining in an ecosystem under current management practices



thus be calculated on any spatial scale for which appropriate data can be gathered or measured. HANPP is applicable on all scales, from plots to municipalities to regions, national territories or the whole biosphere. Note, however, that trade (import/export) is not taken into account, so according to the present definition the HANPP of a country refers to its national territory, not to the consumption taking place within its national economy. In order to improve links to economic activities, e.g. to the activities taking place within a national economy (as measured by GDP), import and export would have to be considered, a task for which reliable methods are presently lacking.

This definition of HANPP is useful for interregional comparisons and time-series analysis. By monitoring HANPP and its various components, such as NPP_{act}, NPP_t, NPP_h, the impacts of different land-use practices on ecosystem energetics as well as their socio-economic performance can be evaluated. Land use may increase or reduce productivity, it may leave more or less energy in the ecosystem, it may yield rich or poor harvests. If agricultural practices succeed in raising NPP_{act}, this results in a decoupling of biomass harvest and HANPP (Krausmann, 2001; Krausmann and Haberl, 2002). This definition of HANPP does not exaggerate human impact by including all NPP of human-dominated ecosystems as appropriated (as some authors have done). HANPP only includes the amount of biomass actually harvested, on top of the NPP prevented by human land use. It is possible to assess HANPP in great spatial detail by combining statistical data with land-cover data derived from remote sensing (Figure 3, Haberl *et al.,* 2001b). In principle, HANPP could be linked consistently to the System of National Accounts (SNA), thus facilitating integrated economic-ecological models of pressures on biodiversity, but actually achieving this goal will require substantial improvements in methods.

HANPP is a measure of the human domination (Vitousek *et al.,* 1997) or colonization (Fischer-Kowalski and Haberl, 1997) of ecosystems. HANPP indicates how intensively a defined area of land is being used in terms of flows of trophic energy in ecosystems (Haberl *et al.,* 2004d). With reference to a given territory, HANPP calculations show how much energy is diverted by humans as compared to the trophic energy potentially available. HANPP is a measure of how strongly human use of a defined land area affects its primary productivity, and how much of the NPP is diverted to human uses and consequently is not available for processes within the ecosystem.

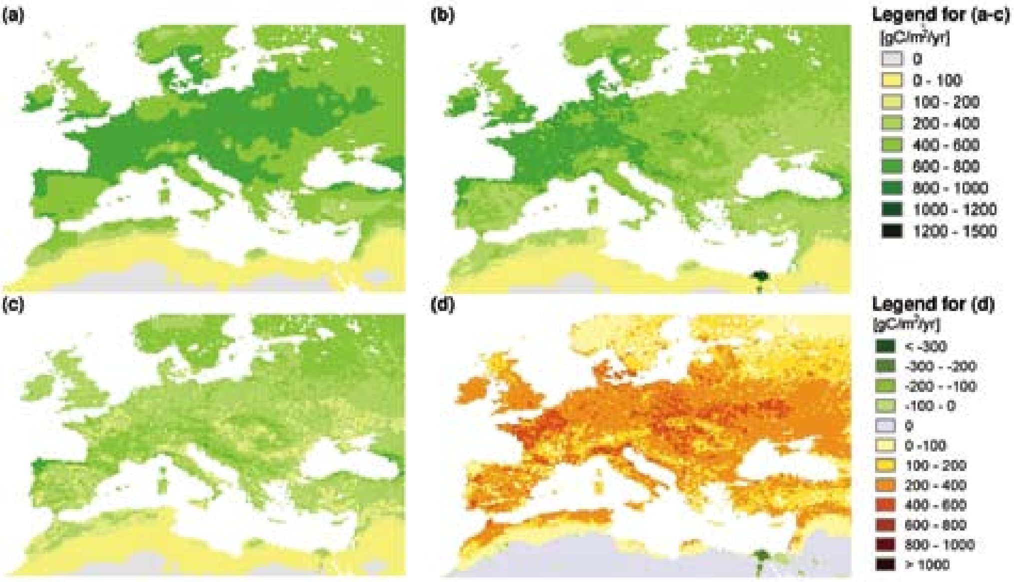
Land use may reduce (e.g. urban settlements, infrastructure, erosion) or increase productivity (e.g. irrigation, fertilization). In arid areas, irrigation may raise productivity considerably above its natural level. HANPP can then become negative, although in many instances it will still be positive, as much of the additional NPP is harvested. For example, Figure 4d shows the Nile delta as an obvious example where NPP_0 is so low that HANPP becomes negative, despite considerable biomass harvest, because of the increase in NPP_{act}.

Trophic energy is one of the most important factors that determines patterns and processes in ecosystems. NPP is the sole energy input of all heterotroph food chains. Many aspects of ecosystem functioning, e.g., nutrient cycling, build-up of organic material in soils or in the aboveground compartment of ecosystems, vitally depend on this energy flow. HANPP demonstrates the impact of human activities on these important ecosystem processes, and thus also on ecosystems services such as carbon sequestration or buffering capacity. Theoretical considerations indicate that a sufficient amount of energy remaining in the ecosystem is necessary for ecosystems to be resilient (Kay *et al.,* 1999). HANPP might impede ecosystem services and thus sustainability: "to the extent that (…) natural systems, species and populations provide goods or services that are essential to the sustainability of human systems, their shrunken base of operations must be a cause of concern" (Vitousek and Lubchenco, 1995, p. 60).

It is plausible that HANPP may be an important driver of biodiversity loss. The theoretical background behind this notion is the species-energy hypothesis (Brown,

FIGURE 4

The human appropriation of NPP in Europe and the Mediterranean around the year 2000. (a) NPP_0, (b) NPP_{act}, (c) NPP_t, (d) HANPP, all expressed in $gC/m^2/yr$. Data were derived using the LPJ model (Sitch *et al.* 2003), a consistent 10x10 km land-use dataset and FAO statistics on agricultural yields and biomass harvest



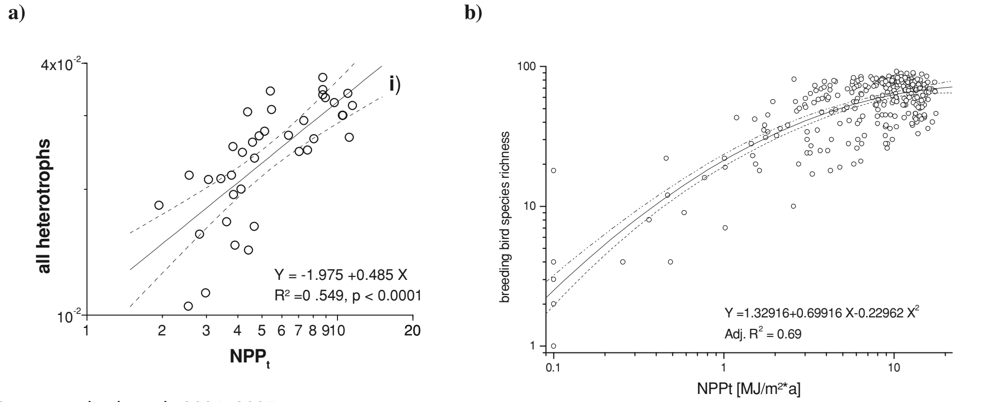
Source: Haberl *et al.* 2006a

1981; Hutchinson, 1959; Wright, 1983) which holds that species numbers in ecosystems depend on the availability of trophic energy. If humans remove energy from ecosystems and lower NPP_t, species numbers would therefore be bound to decline (Wright, 1987; Wright, 1990). On an abstract level this seems obvious. Biomass is the mass of living or dead organisms present in a system. The very idea of trophic-dynamic process in ecosystems (Lindeman, 1942) is an abstract notion for organisms coming into being, growing, and dieing. This process is fuelled by various metabolic processes taking place within organisms. Energy enters organisms above all through two processes: photosynthesis and ingestion of dead or living organisms or parts thereof. Human-induced changes in this process affect patterns (including biodiversity), processes, functions, and services of ecosystems almost by definition.

At present, only indirect tests of the claim that a reduction in NPP_t reduces species richness are possible. As data on potential species richness (S_0) of current landscapes are lacking, there are also no data on the *change* in species richness (ΔS) compared to the potential state. Moreover there is no linear relation between HANPP and NPP_t, the factor that should influence the spatial pattern of current species richness (S_{act}). NPP_t can be low because of high HANPP, but also because of low NPP_0. Without data on ΔS it is therefore not possible to directly test the HANPP/biodiversity relation. *Indirect* tests of HANPP assume that correlations between S_{act} and NPP_t in current, human-dominated landscapes imply that a reduction in NPP_t lowers species richness, which is exactly what was found in two studies. The first study (Haberl *et al.*, 2004c) was based on a transect of 38 squares sized 600x600 m in east Austria. Species numbers of seven taxonomic groups (vascular plants, bryphytes, orthopterans, gastropods, spiders, ants, and ground beetles) were correlated with HANPP and its components. The study found a highly significant positive correlation between NPP_t and species richness ($0.13<r^2<0.76$, depending on taxon). A second study (Haberl *et al.*, 2005) analyzed the interrelations between HANPP and bird species richness in Austria. Some simple measures of land-cover heterogeneity and landscape heterogeneity were also assessed.

FIGURE 5

Correlation analyses between log(NPP_t) and the logarithm of species numbers of various groups. a) NPP_t and "all heterotrophs" (5 taxonomic groups) on 38 east-Austrian plots sized 600x600 m; source: Haberl *et al.* (2004) and additional unpublished data. b) NPP_t (NPP remaining in ecosystems) and bird species number on 328 plots sized 1x1 km randomly selected from Austria's total area of about 83 000 km²; source: Haberl *et al.*, 2005



Source: Haberl *et al.*, 2004, 2005

Four different plot sizes were considered: 0.25x0.25 km, 1x1 km, 4x4 km, and 16x16 km. A nested representative sample of N=328 squares of each size was randomly chosen. The results suggest that NPP variables generally explain bird species richness much better than all available landscape heterogeneity indicators. Consistent with the species-energy hypothesis highly significant, non-linear, positive correlations between NPP_t and bird species numbers were found. Selected results of the two studies are displayed in Figure 5.

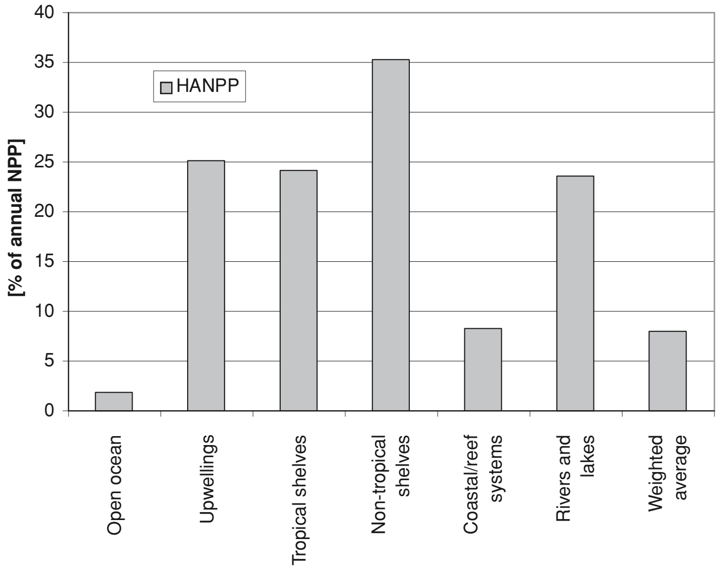
It is possible to apply the HANPP concept to aquatic systems. Indeed, the seminal paper by Vitousek *et al.* (1986) already estimated that global ocean fish catch was 75 million t/yr wet weight in the early 1980s which equals about 20 million t dry matter. Assuming that, on average, fish caught fed on the second trophic level, and assuming 10percent ecological efficiency between levels in food chains, Vitousek *et al.* (1986) estimated the global amount of NPP required to support yearly fish catches to be around 2 000 million t dry matter/yer or 2.2 percent of total aquatic NPP. A later study split global annual fish catches for 1988–1991 into 39 species groups and assigned to these fractional trophic levels, based on trophic models (Pauly and Christensen, 1995). Using again an assumption of 10 percent energy transfer efficiency between trophic levels, this study estimated total primary production required (PPR) to support global fisheries in the late 1980s/early 1990s to be 6 300–14 400 million tonnes dry matter/yr or around 8 percent of total yearly aquatic NPP.[3]

The aggregate global figure of 8 percent seems low compared to the estimates of global terrestrial HANPP of 20-40 percent (Vitousek *et al.,* 1986, Wright 1990; Imhoff *et al.,* 2004, Haberl *et al.,* 2006a), but as Figure 6 shows, the pressure is very unequally distributed to ecosystem types. In open oceans, where most aquatic productivity occurs (about 75 percent), only a small percentage of total NPP ever becomes available to higher trophic levels that could, in principle, be harvested (Pauly and Christensen, 1995). More productive systems which are more suitable for fishing are used more intensively, and most authors agree that current levels of fish harvest have already

[3] Assuming a carbon content of aquatic dry matter biomass of 50percent the figure of 8 percent of 91 800 million tonnes dry matter/yr used by Pauly and Christensen (1995) would equal 3 700 million tonnes C/yr.

FIGURE 6

Yearly NPP required to sustain fish caught per year (average of 1988-1991), by ecosystem type



Source: Pauly and Christensen, 1995

depleted many fish stocks, making marked future increases in fish harvests unlikely (Pauly and Christensen, 1995; Naylor *et al.*, 2000; Pauly *et al.*, 2002; Delgado *et al.*, 2003).

To summarize, this review shows that HANPP has mostly been used to account for the intensity of land use with reference to a defined territory. The contribution of different human uses of the land to total HANPP can be quantified. For example, Table 1 shows that on a global scale around the year 2000, agriculture was responsible for almost three quarters of total terrestrial HANPP. It is more difficult, however, to determine the HANPP caused by a national economy, an economic sector, a defined agricultural activity, or even a defined product. This will require to consistently assign HANPP caused by traded products, an issue that has so far not received sufficient attention in the literature on terrestrial HANPP.

While HANPP has been applied to aquatic systems, its meaning is different in this case, as humans use terrestrial and aquatic systems in different ways (Pauly *et al.*, 2002). In terrestrial systems, purely extractive activities are limited to hunting of unmanaged, wild-living animals, rather small-scale gathering activities of plants or parts thereof, and extraction of timber or other forest products in unmanaged forests. Most biomass, however, comes from more or less intensively managed ecosystems, be they croplands,

TABLE 1

Contribution of different activities to global HANPP in the year 2000 (fishery data refer to 1995)

	Global HANPP	**Contribution to total terrestrial HANPP**
	[000 million tonne C/yr]	[percent]
Cropping	7.56	51.6 percent
Livestock grazing and hay harvest	3.20	21.8 percent
Forestry	1.49	10.2 percent
Infrastructure areas	1.27	8.7 percent
Human-induced fires	1.14	7.8 percent
Global terrestrial total	14.66	100.0 percent
Aquatic HANPP caused by fishery	3.67	

Sources: Haberl *et al.*, 2006a (terrestrial), Pauly and Christensen, 1995 (aquatic) grazing areas or meadows, or managed forests. In aquatic systems, most of the biomass is extracted with little, if any, attempt to manage the system beyond some (often too weak) rules that limit extraction, although the increasing role of aquaculture suggests that this could change in the next decades. Moreover, animals make up the lion's share of the biomass extracted from aquatic systems, whereas plants play only a minor role. This is completely different in terrestrial systems, where plant use is much more prominent, and hunting plays only a minor role in terms of quantity (and is consequently neglected in most HANPP studies). Applying the HANPP concept to aquaculture thus requires new methodological developments discussed in the next section.

POSSIBLE APPLICATIONS TO AQUACULTURE

Issues to be addressed

Sustainability problems associated with aquaculture are manifold, and for some of them, MEFA methods may not be the first choice to address them. For example, escape of farmed organisms is more relevant in terms of genetic changes in wild-living populations than in terms of material flows, although escaping organisms must be regarded as part of the material output or outflow of aquaculture systems. Analyses of quantities of outflows may also be insufficient to capture pollution effects, if they are not complemented by information on chemical quality of these outflows. Adoptions and further developments of the MEFA framework can nevertheless be useful in addressing the following issues:

- Sustainability problems associated to direct and indirect material and energy inputs, above all feed, fossil fuels, industrial materials, etc. and material outputs (wastes, emissions).
- Sustainability problems associated with land demand (in terrestrial-based aquaculture systems), and possibly also those associated with space needed for aquatic systems such as seabed bottom rearing, suspended nets, cages, etc., although its application to the latter category of systems is less straightforward.
- Sustainability problems associated with the appropriation of aquatic biological productivity at different trophic levels.

The following subsections will discuss the potential of existing MEFA methods and the needs to further refine or combine them in order to tackle these issues.

Material and energy flow analysis: applications to aquaculture

Material and energy flow analysis is a systems approach. Its application to aquaculture, as to any other system, therefore hinges on appropriate system definitions, including precise definitions of stocks and flows, and considerations of data availability. The direct inputs and outputs of material or energy, i.e. those flows that cross the boundary of the production system under consideration, are at the heart of any MEFA account. To our knowledge, until now no MFA or EFA that would apply explicit system boundary definitions and aims at covering all inputs and outputs has been carried out for aquaculture systems.[4] Considering past experiences with the application of the MEFA framework to a variety of systems at different scales (villages, cities, economic sectors, companies), we do not expect substantial difficulties here. Given appropriate technical information, both the definition of the production system and the compilation of the databases should be possible. Such classical material or energy flow accounts have, as they measure the flows at their entrance and exit points, a clear conceptual link to the production system in question. This is an important feature of the MEFA framework

[4] Material flow studies of fish farming systems, as the one presented by Brummett (this volume), are extremely important. They do, however, not explicitly address the issues of comparability and standardization. In our opinion, though, explicit system definitions are an essential prerequisite of comparability and therefore of utmost importance for any comparative evaluation.

which enables integrated ecological-economic analyses, by linking economic and biophysical accounting.

To interpret material or energy flows in terms of environmental consequences, though, it is not sufficient to measure the direct material and energetic inputs and outputs of a given socio-economic system, for example an aquaculture production system. To explain why, we use the example of a national economy to which in principle the same problems apply. In contemporary economies, socio-economic systems at any level beyond the global one receive their material and energy inputs not only directly from their natural environment, but also from other socio-economic systems (through imports). Likewise, most socio-economic systems deliver their output not only directly to the environment (as wastes and emissions), but also to other socio-economic systems (as exported goods). This implies, however, that material or energy flow analysis measures input and output flows which represent different stages of the economic production chain. For example, the direct material input as accounted for in national MFA may include the primary extraction of copper ore on domestic territory and sum up this figure with imported copper or even imported copper wires in electric appliances.[5]

Note that both ecological and socio-economic material and energy flows are commonly represented in the form of chains that distinguish different stages. In ecosystems, these include primary producers, consumers at different trophic levels, and decomposers. In an economy we find stages such as extraction, different stages of manufacturing, final consumption, and waste disposal. Obviously, these stages correspond to different system boundaries. What MFA measures as a direct input flow into a socio-economic system may in principle be a flow at any stage of such an ecological or socio-economic material or energy flow chain. In other words, the metabolism of a socio-economic system may be situated between any stages of the ecological and economic production chain.

This has two important implications. The first implication is that a comparison of aggregates of direct input flows needs cautious interpretation (Weisz *et al.*, 2005a; Weisz, Krausmann and Sangkaman, 2006). The second implication is that the flows are probably often not measured at the relevant stage in the ecological or socio-economics production chain. It is commonly accepted to consider only those material and energy flows that cross the boundary between the economy and the natural environmental as causing environmental pressures. For example, CO_2 emissions resulting from respiration of wild-living heterotrophs are not regarded as an environmental pressure, whereas the chemically identical CO_2 emissions from fossil fuel combustion are considered to be environmentally relevant. The theoretical solution would be to combine a systems-based MEFA analysis with an estimation of the upstream or downstream requirements of the direct biophysical flows.

This means to trace back the direct flows to that point in the socio-economic production chain where the extraction from or the release to the environment takes place. The question of how exactly to carry out such estimation leads to a class of problems that is being discussed intensively by material and energy flow analysts today. These problems have, however, been recognized much earlier in both ecology and economics, and the contemporary discussion can greatly profit from these earlier studies. At present two broad classes of methods exist to tackle these issues, (1) those based on an Life Cycle Analysis (LCA) approach, and (2) those using input-output (IO) tables.

LCA starts from the production chain concept and estimates the upstream biophysical requirements (not necessarily restricted to materials and energy; land, water,

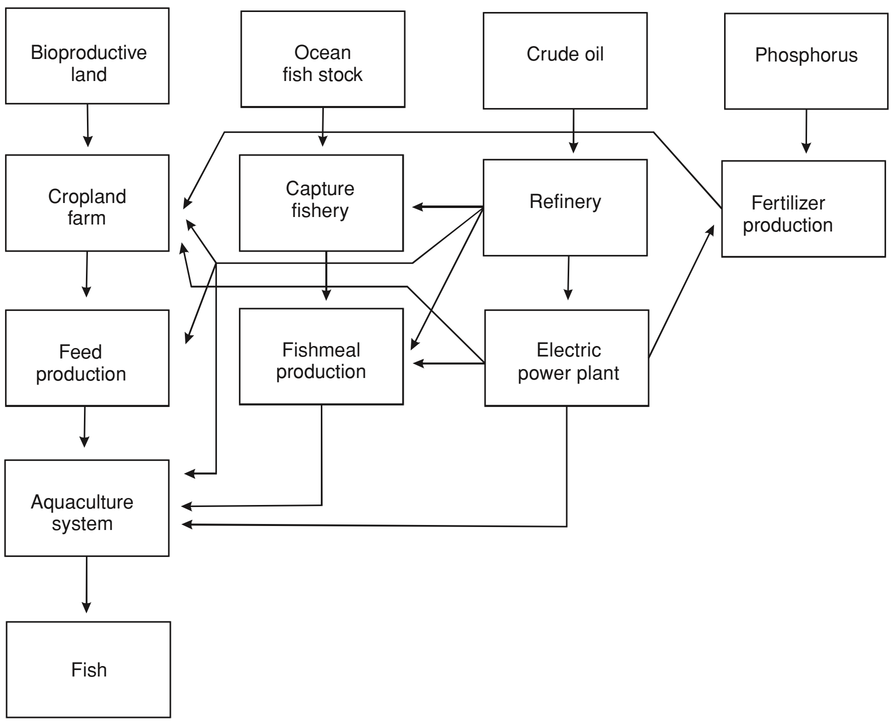
[5] We already mentioned the compatibility to the system of national accounts as one reason why this has become a standard in MFA. Another reason is simply data availability.

and other factors may as well be included) of producing one unit of a final product by use of coefficients. IO methods incorporate the more complex concept of networks and use matrix calculations to track the direct and indirect upstream requirements of materials or energy. Wassily Leontief was rewarded the Nobel price for economics for this innovation. Originally developed in economics (Leontief, 1936), IO models have also played a role in ecology (e.g. Hannon, 1973; Szyrmer and Ulanowicz, 1987).

Irrespective of the numerous technical variants that exist for both LCA and IO models there are some fundamental features which distinguish these two classes of models. Figures 7 and 8 describe the same, hypothetical (made-up), simplified aquaculture system. Figure 7 follows the LCA approach, Figure 8 uses the IO logic, so the differences of the two approaches can be grasped by comparing the two figures.

FIGURE 7

Representation of material and energy flows required for fish production in a hypothetical land-based aquaculture system. Note that the direction of arrows in this figure represents the physical flows; the calculation itself works the other way round, i.e. from final product to primary inputs and is based on technical coefficients. See text for further explanation



The LCA approach takes its start from the product (in our case marketable fish), and asks what was needed to produce a defined amount of this product. In our simplified example we assume fish production in a land-based aquaculture that requires energy in the form of oil-derived fuels and electricity and fish feed based on both cropland agriculture and fishmeal from wild-catch. LCA separately traces back the upstream resource requirements for each of these inputs, up to a predefined primary input stage. These primary inputs are called factors of production or factor inputs in economics. In our example, plant feed production needs a certain amount of plant biomass harvest on cropland. This in turn needs land, fertilizers and energy (for reasons of simplicity we again restrict this to oil products and electricity) for its production. Land is provided by the environment, it therefore represents the environment-economy boundary stage (i.e. a factor of production in economics). Both energy and fertilizers have to be

FIGURE 8

Schema of an Input-Output Table for a hypothetical aquaculture production system. Inputs are in columns (read from bottom to top), outputs in rows (read from left to right). See discussion in the text for explanation

	Aquaculture	Energy sector	Fertilizer production	Plant based fish feed	Final Consumption	Total Output
Aquaculture					x	**x**
Energy sector	x	x	x	**x**		x
Fertilizer production				x		x
Plant based fish feed	x					x
Crude oil extraction		x				
Phosphorus extraction			x			
Fish catch	**x**					
Bioproductive land	x	x	x	**x**		
HANPP	x	x	**x**	x		

produced by the socio-economic system, so their life-cycle chain must be traced back further. In our case, the life-cycle chain of fuels and electricity both go back to crude oil as primary input (assuming a oil-fired thermal power plant) and land as factor input. The life cycle of fertilizers (again simplified) goes back to phosphorus and energy, the latter is going further back again to oil and land.

The overall LCA framework is represented by a multiple bifurcated product chain (Figure 7). Technical coefficients are used for quantification. The advantage of this framework is that it requires relatively few data, compared to the IO approach. The disadvantage, however, is that no consistency checks are built in, and the relations between the separate product chains are not considered (e.g., the use of by-products from one chain as input to another chain). The more complex and the larger the system is, the larger the error margins will be. These shortcomings have also been recognized by the LCA community and initiated attempts to use an IO framework instead or in combination with LCA (e.g. Lave *et al.*, 1995; Suh, 2004).

Figure 8 shows the same production system in an IO framework. Inputs read along the columns from bottom to top and outputs read along the rows from left to right; flows (or factor requirements, respectively) are indicated by an "x". For example, the first (left-hand side) column shows the input structure of the aquaculture system, the first row (top row) shows its output structure. The Figure shows that aquaculture production receives inputs from energy, fertilizer and plant biomass production sectors and delivers its product only to final consumption. As factor requirements aquaculture needs fish (from wild catch), and land. Therefore, a HANPP value can also be attributed to the aquaculture system, even if its direct HANPP (due to land take) may be low or inexistent.

For those not familiar with input-output analysis it is probably not immediately obvious where the decisive difference to LCA is. In the following paragraph we will elaborate this in a non-technical way. First, IO attempts to "express the total direct and indirect flows between any two compartments of a system" (Hannon, 1973). This implies a move from a bifurcated chain perspective to a network perspective expressed by the matrix structure of the IO table. It follows that for any two compartments of the system, the question has to be answered whether there are flows between them, and how large they are. In the schematic example shown in Figures 7 and 8, the LCA framework shows exactly the same connections as the IO framework does. But this is simply our built-in assumption. In real case studies the conversion of a flow diagram into an input output table will often require the consideration of new, so far unrecognized connections.

Moreover, by explicitly distinguishing between intermediate flows, factor inputs and final outputs (represented by the above left, the below left and the above right matrices in Figure 8), IO applies an unambiguous and comprehensive system definition. It follows that unlike LCA-based databases, IO databases (i.e. IO tables) clearly indicate what can be summed together and what cannot be summed together.

Finally, and most important, mathematical algorithms are available for the IO framework to compute the direct and indirect requirements (i.e. requirements via intermediate deliveries) of production factors (e.g. primary material or energetic input, land requirements, HANPP requirements) needed for the production to be allocated to one unit of final product (in our case fish). These algorithms solve the consistency and double counting problems of the LCA approach mentioned above.

One version of this calculation method is known as the Leontief system and is predominantly applied in economics (Leontief, 1941). It represents a demand-driven system, i.e. it assumes that the primary input requirements are determined by final demand. The alternative, known as the Ghosh system, is a supply driven system which assumes that the quantity of the final product is determined by the availability of the primary factor inputs (Ghosh, 1958). Ecological applications of input-output models have always used some variation of the Ghosh model (see Suh, 2005 for a recent review of the comparison between economic and ecological input-output systems). Obviously, also for biologically-based economic production systems, such as fish production, the Ghosh model is more appropriate.

When it comes to evaluation of environmental costs of productions systems, an extension of the MEFA framework (including HANPP) by IO models is in our opinion the method of choice. IO is superior to LCA regarding conceptual reliability and empirical accuracy. The mathematics of the IO models needed are also in place. Data requirements, however, are arguably higher for IO models as compared to LCA models. It will require some real case studies to check the feasibility of such an integrated MEFA-HANPP-IO approach.

Human appropriation of NPP (HANPP): applications to aquaculture

As discussed above in Section 2, HANPP can be used to evaluate ecological impacts of land use, but it has so far not often, if at all, be defined with reference to socio-economic systems, or even more specifically, to defined production systems such as aquaculture. As in the case of material and energy flow accounts, it is essential to distinguish between direct and indirect effects, i.e. HANPP caused directly by the production system (e.g. changes in NPP/biomass flows resulting from a maize field), and HANPP caused by the procurement of inputs (e.g. HANPP caused by corn-based feed used in an aquaculture system).

The application of HANPP to account for ecological pressures arising from land demand of terrestrial-based aquaculture systems is conceptually rather straightforward. A particularly relevant example is the loss of mangrove swamps due to maricultural practices. It is estimated that shrimp, prawn and fish ponds are responsible for 50percent of the loss of mangrove systems in the Philippines and 50percent-80percent in Southeast Asia (Valiela, Bowen and York, 2001). The problem is exacerbated by the short life span of such ponds of only 5-10 years due to eutrophication, accumulation of toxins, sulfide-related acidification, and crop diseases (Valiela, Bowen and York, 2001). The rate of recovery of abandoned ponds is much slower than the rate of conversion of previously untouched mangrove areas to new ponds (Valiela, Bowen and York, 2001). Assessing the HANPP caused directly by such ponds would require the quantification of NPP of untouched mangrove ecosystems, biomass harvested or destroyed in pond construction, NPP of operative ponds, and NPP of abandoned aquaculture systems over time, until the system returns to the original state (if it does so).

Not all of these data seem easy to gather, however. Data from the literature suggest that mangrove systems are quite productive: an older study reported total NPP of a *Rhizophora mangle*-dominated system in southeastern Puerto Rico to be 0.93 kg DM/m^2/yr (Murphy, 1975), a more recent study found above-ground NPP of two mangrove stands in Sri Lanka to be 0.7 and 1.2 kg DM/m^2/yr (Amarasinghe and Balasubramaniam, 1992). An effect that should also be taken into account in this context is that mangroves have a positive effect on the availability of nutrients to adjacent primary producers, e.g. seaweeds or algae, and have been demonstrated to have a positive effect on algal production rates (Koch and Madden, 2001).

If all the above-discussed effects could be quantified, HANPP resulting directly from a shrimp or fish pond over its lifetime could be calculated and should then, for reasons of comparison, be related to its total output over its lifetime. Calculation of direct HANPP effects of other land-based aquaculture systems should be rather straightforward, at least conceptually, and would follow the same logic as the one outlined for shrimps ponds in mangrove ecosystems. It might even be possible to use the same logic also in the case of purely aquatic systems, such as cages, etc., although their effluents might even have a positive effect on the NPP of adjacent water bodies, as they are probably very nutrient rich (this may nevertheless be regarded as ecological detrimental).

In the case of most aquaculture systems, however, indirect effects are much more interesting, particularly those of feed provision. Based on appropriate material flow data it should be possible to evaluate the HANPP caused by the inputs. As discussed above, this would not have to be restricted to inputs derived from land-based systems but could, in principle also be extended to inputs derived from aquatic systems. Several difficulties emerge, however:

- One problem is that inputs needed for a production process such as aquaculture may be derived from various systems located all over the world, which raises two problems. The HANPP per unit of material required, however, depends not only on the material itself, but also on the production system with which it was supplied. For example, the HANPP caused by producing 1 kg of wheat depends on the location of production (productivity of potential vegetation) and on the yield of the cropland system; in addition, losses during transport, processing and storage would also have to be taken into account.
- Aquaculture involves inputs derived from terrestrial and aquatic systems. Although the HANPP approach has been applied to aquatic systems, it has a quite different meaning there, as humans use aquatic systems in a way that is completely different from human use of terrestrial ecosystems. Therefore it is currently not useful to directly compare the results from calculations of HANPP in terrestrial and aquatic systems, and consequently aquatic and terrestrial HANPP should not be added.

Methods developed in the framework of the Ecological Footprint approach may be useful to tackle the first problem. For example, one could use national averages of agricultural yields (Haberl *et al.,* 2001a; Erb, 2004; Wackernagel *et al.,* 2004), nation-specific accounts of the contribution of domestic production and import (Erb, 2004) and national averages of ΔNPP_{LC} on cropland (Haberl *et al.,* 2006a) to estimate the HANPP caused per unit of plant feed in any country. Based on FAO feed balances it would also be feasible to do the same for animal products. This would allow to estimate the amount of terrestrial HANPP caused by the feed used in an aquaculture system, and would at the same time also contribute to evaluating the HANPP caused by terrestrial-based agricultural production systems.

The second problem seems to be more fundamental, as it results from the fact that human use of terrestrial and aquatic systems is so different: While terrestrial systems are actively altered and controlled through application of human, animal and inanimate labour – a process that has been denoted as "colonization of natural processes"

(Fischer-Kowalski and Haberl, 1997; Haberl and Zangerl-Weisz, 1997) – the extraction of resources from aquatic systems through fishing is seldom, if ever, actively controlled or managed. At best, stocks are monitored and harvests limited (Pauly *et al.*, 2002). Therefore, a direct comparison of the amount of dry matter biomass taken from fished aquatic and farmed terrestrial systems is of limited, if any, significance, even if the primary production required (PPR) to sustain the amount of harvested fish is taken into account. One major reason for this is that agriculture can, and does, influence the NPP of terrestrial systems, thus also allowing humans to "decouple" biomass harvest from HANPP to a quite significant extent (Krausmann, 2001). In addition, while it may be possible to sustain a large percentage of HANPP over long periods of time in managed agro-ecosystems, a much smaller relative HANPP figure may result in the depletion of huntable animals stocks.

Another difference between aquatic and terrestrial systems has to do with the level in the food chain at which extraction occurs. The bulk of the biomass gained by humans in terrestrial systems are plants, whereas in aquatic systems humans mostly extract animals, e.g. harvest occurs on another level in the food chain. As already discussed in Section 2, only a limited fraction of the NPP ever enters pelagic food chains, thus eventually supporting fish species further up in the food chain, i.e. the larger ones that can be used commercially. In such cases it might be more sensible to calculate, for example, the "human appropriation of net secondary production" in the case of a fish species that feeds on the first trophic level (and so on for the other trophic levels). On the other hand, such an approach would further complicate comparisons, as results for the different trophic levels could not be summed up, of course.

CONCLUSIONS

We conclude that a combination of material and energy flow accounts with the HANPP concept could contribute important insights in assessing the environmental costs of aquaculture. We have discussed some of the conceptual and methodological challenges to actually use this framework, and are well aware that further work is required in order to realize this potential. In our view, the MEFA framework, including HANPP, should be combined with IO methods to derive accurate, reliable, and double-counting free accounts of the inputs required per unit of product derived from aquaculture systems. The same models and system boundaries should and could also be applied to other agricultural production systems in order to derive indicators that can be directly compared across production systems. The next step would be to apply this concept to a limited number of case studies to test its applicability and real-world feasibility.

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