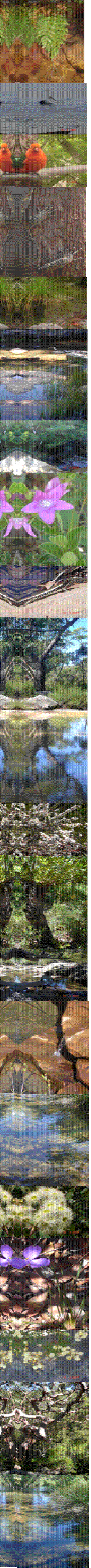

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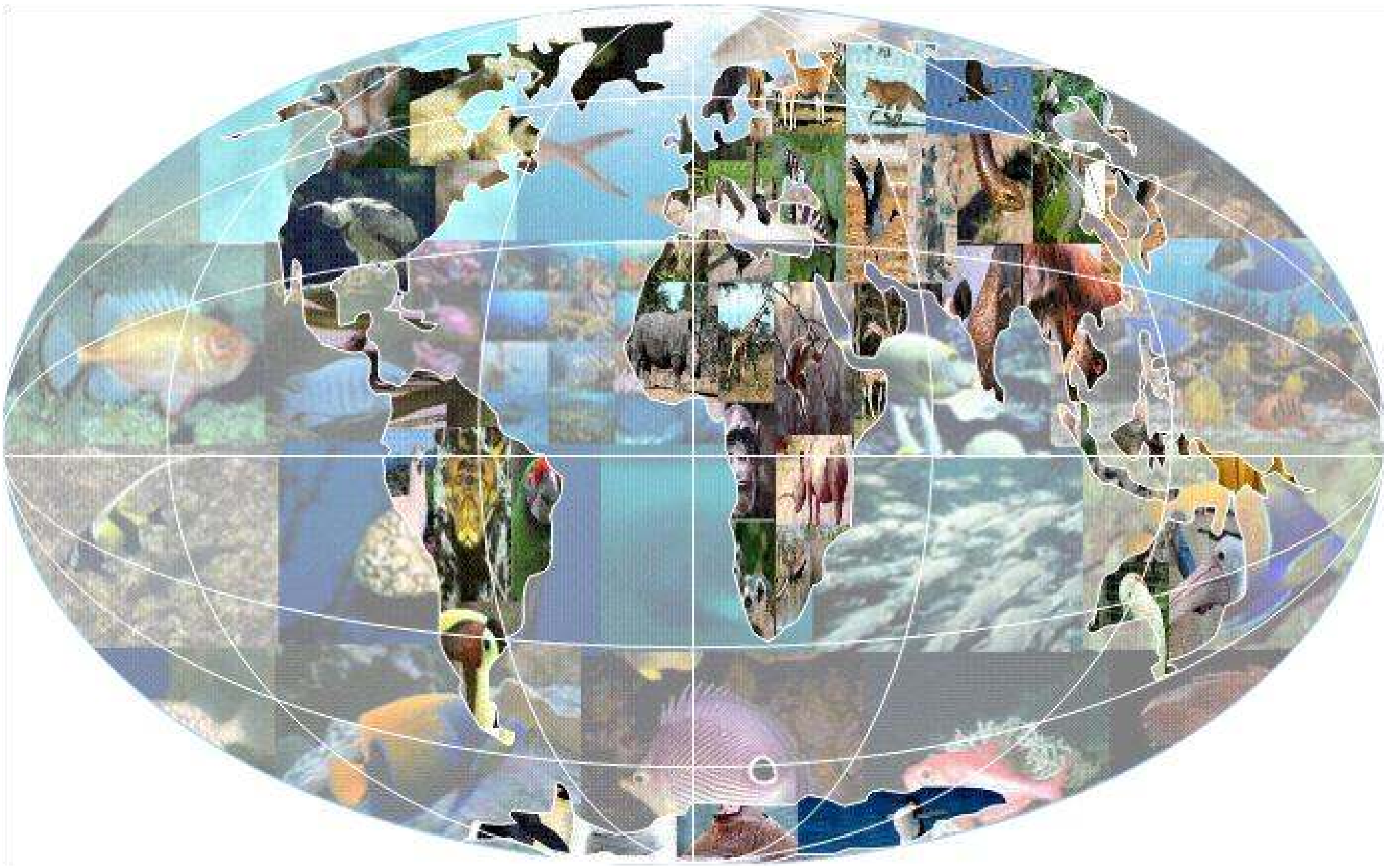
Forecasting the Ecological Footprint of Nations: a blueprint for a dynamic approach



ISA Research Report 07-01 **With a foreword by Mathis Wackernagel**

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Forecasting the Ecological Footprint of Nations: a blueprint for a dynamic approach

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Foreword

I am delighted about Manfred Lenzen and Tommy Wiedmann's initiative to create a Dynamic Ecological Footprint approach. The idea is to equip the Ecological Footprint with techniques that are similar to well-established procedures in the climate debate: climate science needs robust accounts for carbon, temperature, and atmospheric carbon concentration to document outcome; and such outcome measures need to be complemented with dynamic climate models to find out what these trends might mean for the future wellbeing of humanity and the biosphere as a whole.

There is no doubt: we too, in the Footprint world, need complementary tools to the static Ecological Footprint accounts. We need tools that can explore how past trends and human interactions with the biosphere might shape our future biocapacity and Footprints. This is what the Dynamic Footprint attempts to do.

I welcome this innovative work and look forward to others joining the debate. Dynamic Footprints are by nature more speculative than *ex-post* accounts like the static Footprint. But as in the aforementioned climate models, they are needed for spelling out the implications of current actions. Further, as more of these models emerge, they can be tested against each other. This will strengthen the science of understanding present and future biocapacity constraints. And possibly most importantly, Dynamic Ecological Footprint models such as this very first one open the debate we desperately need: how can we plot a course that leads to high quality of life for all within the means of our one planet?

Warmest wishes

Mathis Wackernagel

Mathis Wackernagel
Executive Director
Global Footprint Network

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Executive Summary

This report provides the theoretical base and an example for expanding the static Ecological Footprint accounting method into a dynamic forecasting framework which is forward looking to 2050, incorporating biodiversity amongst other factors, into a causal network of driving forces, and taking into account globalised trade with its complex supply chains. It achieves these objectives by applying state-of-the-art numerical techniques such as multi-region input-output analysis, finite-difference temporal iteration, spatial autocorrelation analysis, Monte-Carlo simulation, Structural Decomposition Analysis and Structural Path Analysis.

This world-first *Dynamic Ecological Footprint* connects the original Ecological Footprint method with some modifications that were anchored at different points of the causal network, such as land use and disturbance, species diversity, and pollution. It thus demonstrates an effective and elegant means for unifying a range of methodologies and objectives into on framework while retaining the research question and metric of the original approach.

As with the static method, the dynamic method measures the amount of biotic resources needed to meet humankind's demand of food, timber etc, and the response to humanity's greenhouse gas emissions. The quantity central to the Ecological Footprint concept is *bioproductivity*, which is the output of renewable biotic resources harvested by humans. The world economy is exerting pressure on the globe's bioproductivity as human population and economic activity continue to expand. Thus, the global Ecological Footprint continues to grow, with the difference between the globe's inherent productivity (its *biocapacity*) and humankind's Ecological Footprint becoming narrower each year.

The results of this first 'blueprint' analysis are intriguing, particularly in three areas. First, the convergence of Ecological Footprint and biocapacity trajectories confirms the World Wide Fund for Nature's *Living Planet Report 2006* which suggests that – in the long term – humankind's demands have been exceeding the world's biocapacity since 1980. We estimate that population increase and economic growth are likely to halve the gap between the Ecological Footprint and biocapacity within this century. This parallels dour prognoses for greenhouse emissions and climate change.

Second, the combined effects of negative drivers of the biocapacity-Ecological-Footprint gap, such as population growth, affluence growth, land degradation and biodiversity decline, are projected to be twice those of positive drivers, such as improved yields and carbon dioxide fertilisation. Amongst the negative drivers, biodiversity decline is poised to become the strongest influence by 2050 in reducing biocapacity, thus once again reinforcing the message of the *Living Planet Report*.

The third result shows that consumption in high-income countries is the most significant driver, through world trade, for both Ecological Footprints and rates of closing the gap between biocapacity and the Ecological Footprint in low-income countries. For example while the Ecological Footprint of the consumption of Canada, the Russian Federation, USA, Sweden and many other European nations is increasing, they are able to maintain a constant remainder of available biocapacity, because they trade and purchase biocapacity from countries such as Brazil, Congo, Venezuela, Australia, Indonesia, Malaysia, Ecuador and Gabon, where remaining biocapacity is rapidly shrinking.

While this study has many caveats, better data and improved equations may not qualitatively change the prime conclusion: that population growth, economic growth, and world trade are literally consuming the living fabric of the earth. In the global climate change conundrum driven by greenhouse gas emissions, these driving forces are still to be acknowledged and constrained.

1 Introduction

Since the Ecological Footprint was invented (Rees 1992; Wackernagel 1994) it has experienced tremendous success in communicating the concept of limited world resources to governments and the general public alike. The Ecological Footprint measures – in terms of *bioproductivity* – the amount of biotic resources needed to meet humanity's demand for food, timber etc, and to compensate for humanity's CO_2 emissions from energy use. This bioproductivity is expressed in "global hectares", representing an area of world-average biological productivity, including both land and water. Global hectares are calculated from actual hectares by weighting with yield factors and equivalence factors (Wackernagel *et al.* 2005). A variety of modifications have been suggested.[1] The Ecological Footprint method has been and is still subject to ongoing discussion, the details of which have been published previously.[2] A standardisation process is currently underway in order to unify diverging methodologies.

1.1 The need for a dynamic Ecological Footprint method

There is substantial evidence that in the long term, diminished ecosystem functioning will deteriorate the services humans are able to derive from natural and artificial landscapes, for example agricultural bioproductivity. According to the literature, ecosystem functioning, in turn, is influenced by a multitude of impact pathways, amongst which the perhaps more prominent involve land use and conversion, and climate change (see Pimentel *et al.* 1976; Naeem *et al.* 1999; Armsworth *et al.* 2004; Asner *et al.* 2004; Spangenberg 2007, Fig. 1).

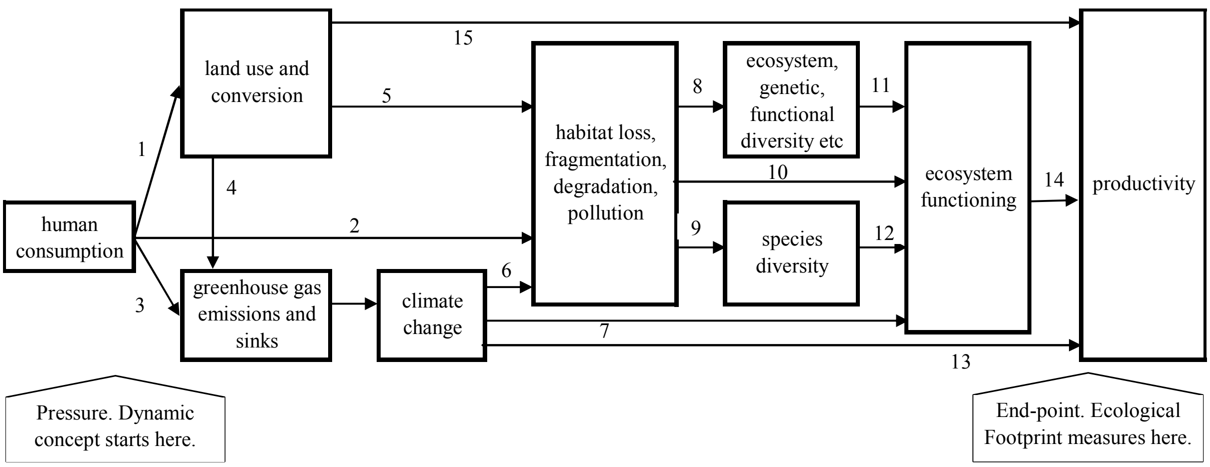

Fig. 1: Impact pathways linking human consumption and agricultural bioproductivity.

[1] Bicknell *et al.* 1998; Simmons *et al.* 2000; Ferng 2001; Luck *et al.* 2001; Ferng 2002; Stöglehner 2003; Ferng 2005; Venetoulis and Talberth 2006; Venetoulis and Talberth 2007 and Lenzen and Murray 2001; 2003; McDonald and Patterson 2003; Chen and Chen 2006; Gao *et al.* 2006; Chen and Chen 2007; Peters *et al.* 2007.
[2] Levett 1998; van den Bergh and Verbruggen 1999; Ayres 2000; Costanza 2000; Opschoor 2000; Rapport 2000; van Kooten and Bulte 2000; Lenzen *et al.* 2007a; Wiedmann and Lenzen 2007.

Apart from the obvious links (1 and 3) these pathways represent[3]

2. the emission of pollutants into soil, water and air, including fertilisers,
4. the emission of greenhouse gases due to land use changes (for example clearing),
5. the fragmentation and degradation of habitat as a result of conversion of land for human purposes,
6. the disappearance of habitat because of changing climatic conditions,
7. the ceasing of certain ecosystem functions because of climatic changes,
8. the decrease and loss of ecosystem diversity because of habitat loss,
9. the decrease and loss of species diversity because of habitat loss,
10. the ceasing of certain ecosystem functions because of the disappearance of the very ecosystem,
11. and 12. diminished ecosystem functioning because of decreased ecosystem and species diversity,
13. changes in bioproductivity as a direct consequence of climate change,
14. changes in bioproductivity as a direct consequence of biodiversity changes, and
15. changes in bioproductivity as a direct consequence of agricultural practices, for example through salinisation, nutrification, eutrophication, or erosion.

In this view, human consumption exerts the *pressure*, acting via land use and greenhouse gas emissions to biodiversity and ecosystem functioning, with bioproductivity being the end-point of impacts, or *state*. To close the loop, available bioproductivity (biocapacity) will in turn limit what humans can consume.

It appears that land use and conversion, and biodiversity constitute drivers of delayed changes in ecosystem functioning, just as emissions constitute a driver for delayed global atmospheric temperature and sea levels. In other words, today's biodiversity and habitat loss will have a profound effect on future bioproductivity levels. In both cases, the relationships are likely to be highly non-linear and may involve threshold effects.

Research that explores and understands these causal links is needed in order to make informed decisions for a sustainable future.[4] Consider the analogy of climate change: If governments only monitored, and acted on signs for climate change (for example temperature levels) in policy making, rather than on greenhouse gas emissions, decision-makers would just be starting to suspect potentially dangerous processes, since anthropogenic influences on

[3] Note that Fig. 1 serves solely to explain the motivation of this work, and is in no way the only and the most general representation of these causal links. Moreover, there are certainly links which are not shown in Fig. 1 such as the loss of species as a direct consequence of changed climatic conditions, that is without involving habitat loss, and the decrease of bioproductivity as a direct consequence of land degradation, without involving ecosystem functioning.

[4] Arrow *et al.* 1995 (p. 93) makes this point for the example of ecosystem resilience: "A more useful index of environmental sustainability is ecosystem resilience. One way of thinking about resilience is to focus on ecosystem dynamics […]. Even though ecosystem resilience is difficult to measure and even though it varies from system to system and from one kind of disturbance to another, it may be possible to identify indicators and early-warning signals of environmental stress. For example, the diversity of organisms or the heterogeneity of ecological functions have been suggested as signals of ecosystem resilience. […] the signals that do exist are often not observed, or are wrongly interpreted, or are not part of the incentive structure of societies. This is due to ignorance about the dynamic effects of changes in ecosystem variables. […] The development of appropriate institutions depends, among other things, on understanding ecosystem dynamics and on relying on appropriate indicators of change. Above all, given the fundamental uncertainties about the nature of ecosystem dynamics and the dramatic consequences we would face if we were to guess wrong, it is necessary that we act in a precautionary way so as to maintain the diversity and resilience of ecosystems."

global temperatures and sea levels have only been acknowledged relatively recently. We know today that it is already too late to avoid long-term climate change per se, however it is the knowledge that *emissions drive temperature levels* that alarmed people about potential climate change more than twenty years ago. Therefore, in terms of Fig. 1, policy needs to investigate "early-warning" drivers that affect bioproductivity.

1.2 Aim of this work

The previous section points to the following issues that need to be addressed:

- The static Ecological Footprint method measures the end-point of the causal chain in Fig. 1 (bioproductivity). It is backward-looking accounting of what occurred, not an extrapolation of how it could affect the future. It thus does not contain an "early-warning" signal: By the time bioproductivity has decreased because of biodiversity loss, habitat loss, and/or land and soil degradation due to unsustainable agricultural practices, it may be too late for abatement action. Therefore, policy needs models that deal with the pressure variables in Fig. 1, and link them to the bioproductivity end-point. [5]
- The static Ecological Footprint method examines a resource question (ie how much bioproductivity do we have and how much do we use?) without asking about ecological or other driving forces that ultimately support bioproductivity. Pursuing the resource question in isolation from ecological factors can lead to outcomes that actually *deteriorate* ecosystems (Lenzen *et al.* 2007a).[6] This is because taking our resource base as a yardstick provides incentives for expanding high-yield croplands and monocultures at the expense of natural ecosystems. Incorporating ecological variables into the Ecological Footprint method avoids these counter-productive incentives.
- The static Ecological Footprint method lumps together two components: land and greenhouse gas emissions. Both quantities are associated with impacts that have significantly different lifetimes: While land and ecosystems may recover or be restored over decades after an initial disturbance, greenhouse gas emissions will have an effect for centuries. As a result, entities abating the long-lived component earlier will cause less future impacts. This circumstance has significant implications for policy and negotiations about sharing the burden of climate change (Lenzen *et al.* 2004b): Ignoring temporal issues can lead to serious distortions in allocations of Footprints to national or sub-national entities (such as companies).[7] A dynamic, temporally explicit method can overcome such distortions.

Thus, this work has the broader aim of attempting a general outline of a dynamic Ecological Footprint method for forecasting and policy analysis that can become a complementary tool to the existing method. More specifically we propose to:

[5] The works of McDonald and Patterson 2003, Lenzen and Murray 2001; 2003 and Peters *et al.* 2007 are predecessors to the dynamic framework presented in this work as they cover pathways 1-6 in Fig. 1. The dynamic method connects these approaches to a common end-point (bioproductivity).
[6] Also at www.isa.org.usyd.edu.au/publications/documents/ISA&WWF_Bioproductivity&LandDisturbance.pdf.
[7] This has been amply demonstrated for the case of climate change: CH_4 and CO_2 have different atmospheric lifetimes, and the effect that abating entities have on the future climate does not only depend on the amounts of emissions reduced, but on the temporal profile of reductions (Rosa and Schaeffer 1995; Rosa and Ribeiro 2001; Rosa *et al.* 2004).

a) incorporate biodiversity variables as driving factors of future Ecological Footprints,
b) shift the point of data input towards the pressure variables in Fig. 1, and link a quantitative analysis through to the end-point bioproductivity.

Thus, the bioproductivity end-point (referred to as the "Ecological Footprint") coincides with the metric of the static method, but it is influenced by driver variables in a dynamic way.[8]

The description of our analysis will proceed as follows: we pick out a number of the pathway nodes in Fig. 1, and in Section 2 we review the quantitative estimates that have been attached to their links. In Section 3 we then venture a bold attempt at finding the "best" compromise of a country-level set of quantitative parameters for these links.

In Section 4.2 we apply these in a temporal analysis of country-level consumption, production, land use, greenhouse gas emissions, species diversity, and bioproductivity up to 2050. Section 4.3 quantitatively decomposes the global trend of narrowing the biocapacity-Ecological-Footprint gap into accelerating and retarding driving forces. Section 4.4 qualifies these results by a comprehensive appraisal of uncertainties. Section 4.5 provides a practical example for how the dynamic method may be applied to sub-national entities such as corporations. In Section 4.6 we present country accounts rankings in terms of the end-point (Ecological Footprint and biocapacity) for 2050. These accounts and rankings thus complement existing static country-level Ecological Footprint accounts (Loh 2006).

In Section 5 we list shortcomings that necessarily arise for a forward-looking dynamic method. Section 6 concludes and provides an outlook.

[8] Using the IMAGE land-use and land-cover model, van Vuuren and Bouwman 2005 already presented a dynamic Ecological Footprint forecast until 2050. However these authors do not cover biodiversity and degradation effects, which are at the heart of this analysis.

2 Interactions of consumption, land use, greenhouse gas emissions, biodiversity and bioproductivity – a literature review

2.1 Human consumption and land use

The amount of land humans require depends on a number of factors, of which the more important are probably 1) population, 2) living standards (affluence), 3) agricultural production, and 4) agricultural yields. More precisely, land use L can be written as a function of population P, gross economic output per capita (x, \$/cap), agricultural production per gross economic output (o, tonnes/\$PPP), and productivity (p, tonnes/hectare) as $L = P\ x\ o\ /\ p$.

2.1.1 Population and affluence growth

A growing population certainly exerts a pressure to increase land occupation, especially when this population is striving towards higher living standards at the same time. The latter is reflected in the fact that growth rates of per-capita real GDP have not shown any remarkable trends since 1970, but rather fluctuated around a mean of about +1.5% per year (Fig. 2). Regressing a 35-year history, growth rates can be modeled to grow for North America (+0.013% year^{-1}), Africa (+0.013% year^{-1}) and Oceania (+0.024% year^{-1}), and decrease for South America (–0.063% year^{-1}), Europe (–0.053% year^{-1}) and Asia (–0.011% year^{-1}).[9] Based on these trends, per-capita and total real GDP (net both inflation) is set to grow $2^1/_2$-fold until 2050 (Fig. 3, compare with forecasts in Tilman *et al.* 2001a, p. 282).

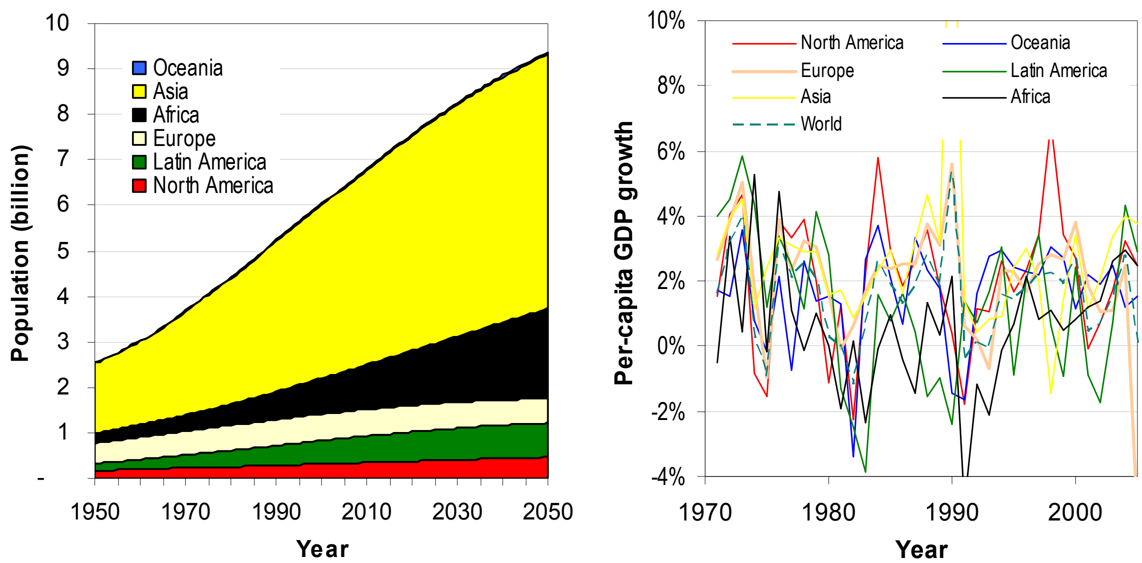

Fig. 2: History and forecast of world population (left, compiled after U.S. Census Bureau 2006), and history of per-capita GDP growth (right, compiled after United Nations Statistics Division 2007b)

[9] Regressing only the period 1990 to 2005 confirms the magnitude of all trends except for South America, which has experienced a slight growth of growth rates during the past 15 years.

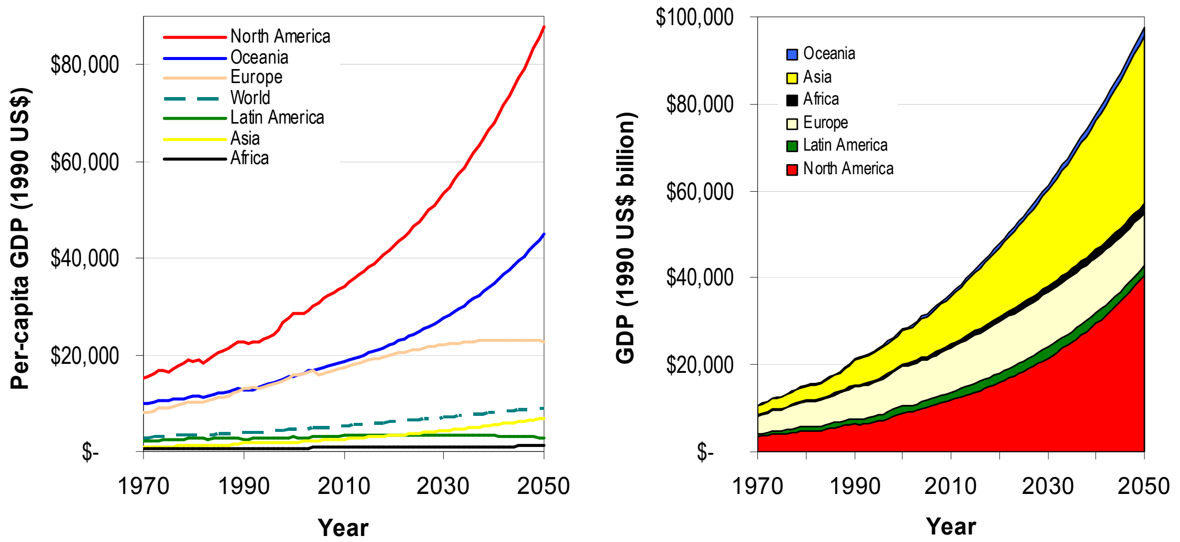

Fig. 3: History and forecast of per-capita real GDP (left) and total real GDP (right, net of inflation, compiled after United Nations Statistics Division 2007b).

2.1.2 Agricultural output

The real GDP growth in the right-hand graph in Fig. 3 forms a strong driver for resource requirements, amongst which is agricultural production. On one hand, agricultural output (in tonnes, say) can be expected to grow proportionally with population. On the other, agricultural output will not follow per-capita GDP: As societies become more affluent, the consumer basket changes to incorporate a higher proportion of manufactured goods and services, both of which require only minor agricultural output (Figs. 4 and 5).

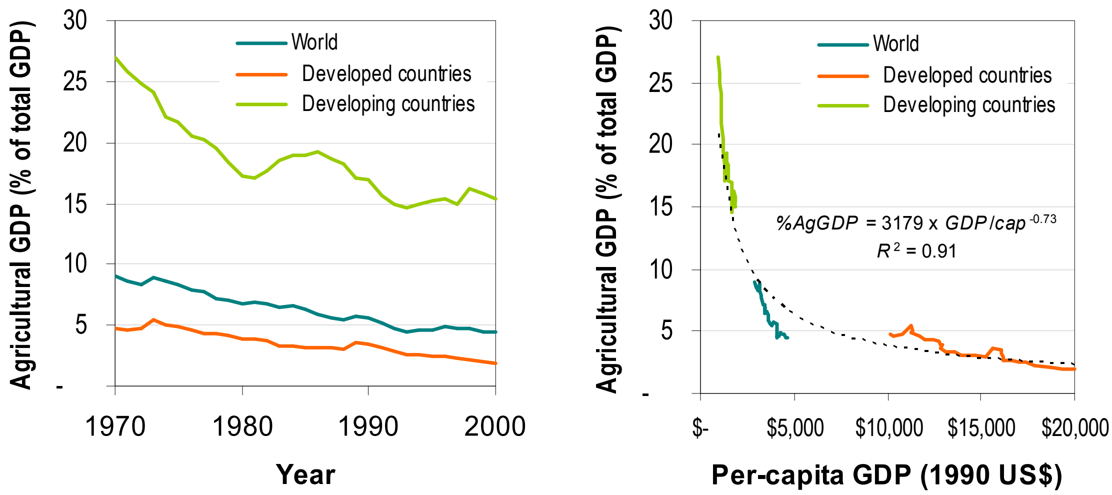

Fig. 4: Agricultural GDP as a percentage of total GDP, as a time trend (left) and an affluence trend (right). Compiled after FAO Statistics Division (FAOSTAT) 2007a.

In fact, during the past 35 years, agricultural GDP as a percentage of total GDP has decreased in all world regions, which can be seen as a time trend as well as an affluence trend (Fig. 4).

A similar trend can be observed from micro-level panel data referring to just one year and one country: The intensity of agricultural and forestry output per unit of household consumption can be calculated for different income classes (Fig. 5). As with the global data in Fig. 4, the Australian panel data shows a marked decrease of the material intensity of the commodity basket as consumers become more affluent.

This material intensity can be analysed using the concept of elasticity. The elasticity η_y of the per-capita material requirement M with respect to the per-capita expenditure on consumer items y for example is defined as $\eta_y = \frac{\partial M/\partial y}{M/y}$. A value of η_y = 0.9, for example, means that, for a 100% increase in expenditure, the material requirement increases by only 90%. When expressed using the elasticity concept, the material requirement can be written as $M = M^{(y=1)} y^{\eta_y}$, and the material intensity is $m = \frac{M}{y} = M^{(y=1)} y^{\eta_y - 1}$. Once again, a value of η_y = 0.9, for example, means that, for a 100% increase in expenditure, the material intensity decreases by $\eta_y - 1 = -10\%$.

Comparing Figs. 4 and 5 shows that the elasticities are of the time series and panel data analyses are quite similar at $\eta_y - 1 \approx -0.7$, or $\eta_y \approx 0.3$. Hence, for every 10% increase in per-capita GDP, agricultural output has to increase by only 3%, or in other words, demand for agricultural output is relatively inelastic. This is not surprising since food is a necessity that saturates at relatively low incomes.[10] In fact, applying a 3% elasticity to Europe's forecast per-capita income and population growth puts 2050 agricultural output at less than 1% over that of 2005, which is in reasonable agreement with the -20% – +30% range estimated by Rounsevell *et al.* 2005.

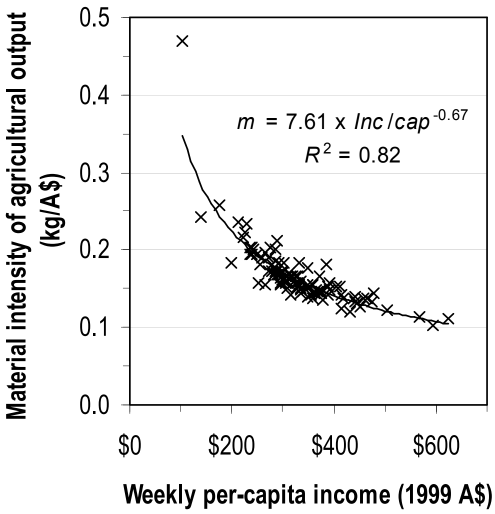

Fig. 5: Material intensity of agricultural output as a function of weekly per-capita income across Australia. Compiled by applying generalised input-output analysis (Australian Bureau of Statistics 2004; Lenzen *et al.* 2006) to Australian agricultural production statistics (Australian Bureau of Agricultural and Resource Economics 2004), and then to Australian data on household expenditure (Australian Bureau of Statistics 2000).

[10] For more details on this type of calculation, see Wier *et al.* 2001; Lenzen *et al.* 2004a; Lenzen *et al.* 2006.

2.1.3 Yield

In a final step, we examine the relationship between agricultural output and land requirements. During the past decades, productivity (yields) in all agricultural systems has increased worldwide as a result of improvements of agricultural techniques, fertiliser use, and other intensification trends (Fig. 6). We have fitted non-linear saturating curves in order to indicate limits to yield improvements. These limits are due to complex relationships between the livestock and crop systems, which we will examine in the following two Sub-sections. It is clear that – in light of these complex causal patterns – our elasticity representation is simplified. Note that our forecast of about 8% productivity increase between 2005 and 2050 is more conservative than the estimate of 20% - 80% by Rounsevell *et al.* 2005. We comment further on this discrepancy in Section 4.2.1.

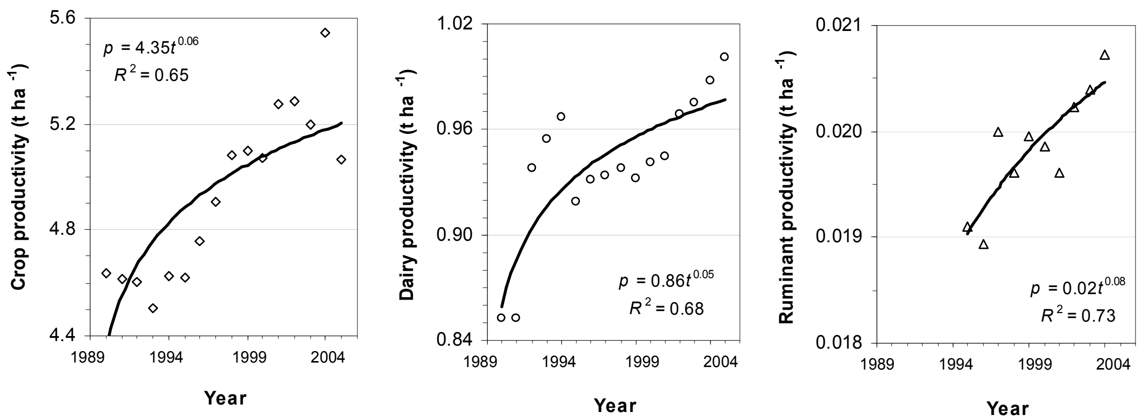

Fig. 6: World crop, dairy and ruminant livestock yields over time. Compiled from country-level data obtained from FAO Statistics Division (FAOSTAT) 2007b and Livestock Information and Policy Branch 2007.

2.1.3.1 Livestock systems

Global animal production is composed of two distinct parts: a ruminant system of cows, sheep and goats which directly occupies land, and the monogastric system of pigs and poultry with large indirect land use through the global trade flows of animal fodder. The two sectors are depicted well in Fig. 7, where ruminants (which are effectively land- and climate-limited) have experienced decreasing production, while monogastrics production has increased. The latter increase has led to intensified food chains, involving more grain and often animal-protein supplements.[11]

The increase in monogastric production or the 'next food revolution' (Delgado *et al.* 2001) has been driven by population and income growth particularly in Asia where intensive animal systems, both at smallholder level and large industrialised food complexes, fit well into peri-

[11] The dairy and cattle feedlot sectors are somewhat in between, as high tech dairying and finishing beef cattle requires supplement food mostly derived from crop concentrates.

urban areas around large cities. The larger scale systems mostly rely on animal feeds imported from cheap production zones. The current focus is on soy production from Brazil and Argentina[12], but intensive animal production has long used fishmeal (Kristofersson and Anderson 2006) for example as a protein supplement, so indirectly linking land-based production systems with decline of regional fisheries. While countries with low population densities can support intensive animal production with their own feed resources, many developed and developing countries import large volumes, particularly of protein rich grains sourced from the cheapest and largest suppliers.

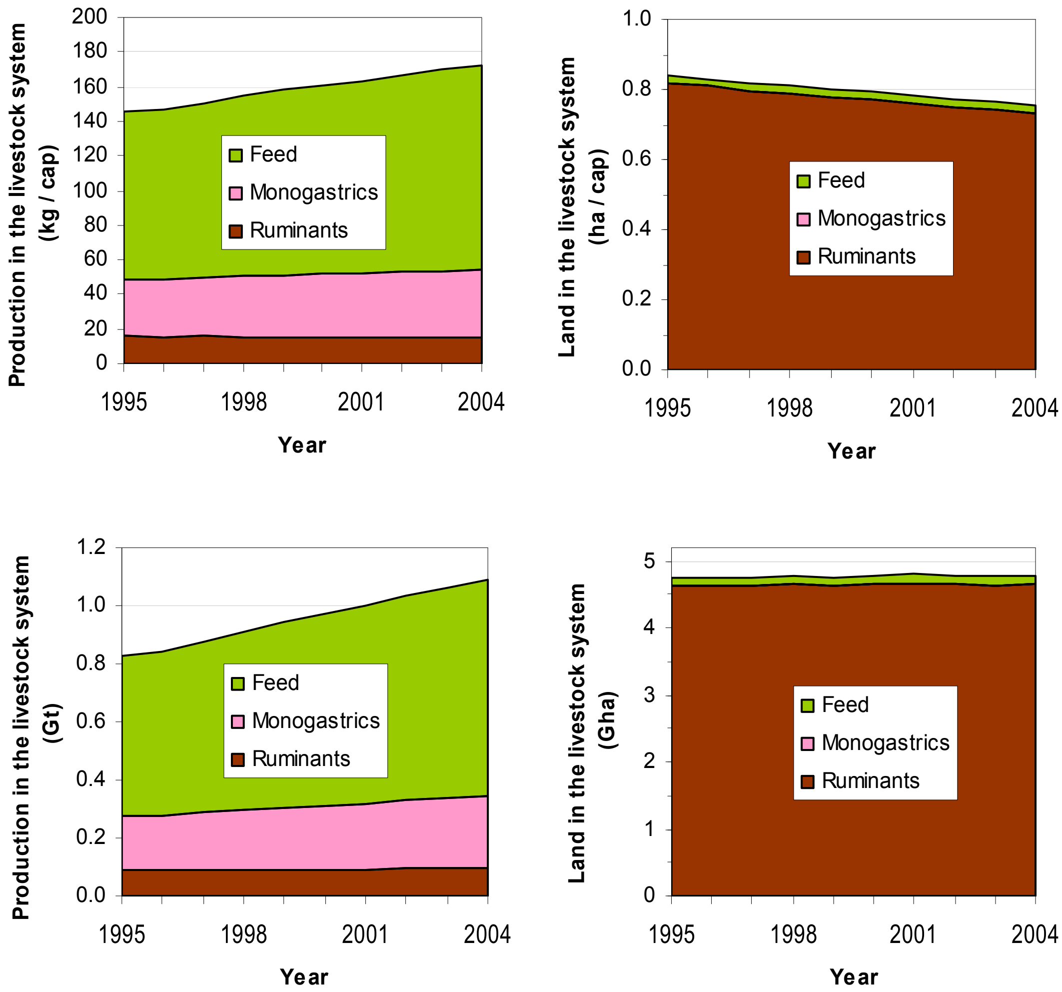

Fig. 7: Production and land in the world's livestock system. Compiled from country-level data obtained from FAO Statistics Division (FAOSTAT) 2007b and Livestock Information and Policy Branch 2007. Mass values for concentrate/grain feed were calculated using a uniform feed-to-livestock (live weight) ratio of 3:1, and added to livestock values (Allameh *et al.* 2005; Reynolds and O'Doherty 2006). Area values were then calculated from the mass values using crop and grazing yields.

[12] Foreign Agricultural Service 2004; Pengue 2004.

Considering that ruminant animals are now land-limited, and monogastric animals rely on cropland often far removed from the source of supply, it is reasonable to expect that technical efficiencies of animal production (meat produced per unit of food eaten) will continue to improve (Bradford 1999). This includes the refinement of animal genetics, better organisation of the full production chain, animal health and so on. However, there are logistical and social limits which could see global productivity tending to saturate perhaps by the decade 2030 when global food production, though not potential, might stabilise due to difficulties with organisation and politics, as well as the ability of arable lands to maintain increasing grain supplies.

The world can probably expect that potential yield of grain crops (the germplasm potential rather than in-field actual production) that underpin monogastric meat production will continue to grow out to 2050 and beyond. Thus monogastric meat production should not be physically limited, although social and political constraints may bite[13]. Total global grain production has tripled in the last 60 years to reach 2,000 million tonnes in 2006. In that period, the per capita grain production has oscillated between 300 and 350 kilograms per person. However the year 2006 saw the world with grain stocks buffer of less than 60 days of world consumption, the lowest it had been since the early 1970s. This probably reflects an organisational issue rather than a biophysical tipping point. However it is wise to be cautious in expecting that the technological supremacy of the last half century will continue unchecked for the next half century.

Four significant uncertainties exist for the continuing expansion of global meat production. The first is the emergence of human diseases linked to intensive animal production systems, which influences willingness to consume animal products. The most recent example is bird flu[14] which regularly requires the slaughter of large populations of poultry and exclusion policies to quarantine the affected region and even country from trading activities. The second is constraints in supplies of oil and natural gas which are expected to peak before 2020 (Robert *et al.* 2006; Government Accountability Office 2007). Apart from breaks in transport and distributional chains, the central issue is nitrogen fertiliser currently manufactured from cheap and plentiful natural gas. Without cheap fertiliser, intensive animal production chains currently fed by cheap grains, will reduce their productivity and human diets may have to become more vegetarian. If issues one and two combine with a series of global economic shocks and regional meltdowns the third issue may develop, where the increasingly integrated system of globalised trade may lose its just-in-time fluency, and become more brittle and erratic. The overarching fourth issue is that of global change which may advantage production systems in northern latitudes while disadvantaging equatorial and southern latitudes.

These issues could affect land use and biodiversity impacts in two ways. One could be a reduction in the animal proportions of human diets, a reduction in global flows of animal feeds and a reduced footprint associated with changed human diets. As global markets for meat decline, the second medium-term option might see longer-lived grazing ruminants retained longer, thereby maintaining pressures until the grazing system crashes and then starts the slow process of refurbishment.

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[13] The social limits are well documented in Peter Singer's book *The Way We Eat; Why Our Food Choices Matter* (Singer and Mason 2006, p. 328 ff) where populations will no longer accept many processes that lie behind the intensive animal production chain.

[14] World Organisation for Animal Health 2007.

2.1.3.2 Crop systems

Modern economies are based on the belief of continuing technical progress and especially in the agricultural sector where the well publicised 'green revolution' led to large increases in grain production for key staples such as rice, wheat and maize. Figure 6 shows the continuing echo of the green revolution with rising yields per unit area of over the last quarter century. A common misconception is that the green revolution was a virtual 'brain-led' one. In reality, the revolution was based on substantial changes to the whole farming system with alterations to plant genetics (the brain part) which required additions of pesticides, fertilisers, irrigation and improved management practices for the improved genetics to show their potential. Some literature suggests that global agriculture is failing to maintain growth rates that match population and affluence growth rates[15] and that a "second green revolution" is required (Wollenweber *et al.* 2005).

Most insiders in agriculture and plant protection are optimistic that food production will maintain pace with rising food demand (Rosegrant and Ringler 1997), and that if regional options are constrained, then global trade flows will help. Supporting the positive case are recent studies projecting a 50% increase in yield of 23 crop types by 2050 (Balmford *et al.* 2005), thus allowing human food supplies to be maintained as population climbs to about nine billion by 2050, and then starts to decline due to the demographic transition and ageing. In the developed world, particularly Europe and North America, there are large areas of land 'set aside' to reduce overall production levels. Additionally rising global temperatures will bring many northern latitude lands into advantaged plant production windows. South America still has significant areas of reasonable soil, which if developed, will underpin part of global grain supplies.

However on the negative size there is a litany of issues which challenge a more optimistic future. Two non-land issues are that of organisation (the ability to supply food easily to people who need it) and trade fragilities (dumping of surplus food and destroying local supply chains). Three important land issues, not often brought into optimistic deliberations, are as follows. The first is over-harvesting of plant production resources where south and central Asia use over 70% of net primary productivity (Imhoff *et al.* 2004), thus limiting the fraction left to preserve ecosystem function. Many irrigation areas that form the strong motors of global grain production have a declining water resource (Global International Waters Assessment 2006) and soil salinisation. Thirdly, global change will negatively affect crop yields in many equatorial regions, at best, adding stress to world trade arrangements, or at worst increasing food scarcity. Dominating all of these issues is the supply of cheap nitrogen fertiliser made from natural gas. As gas supplies dwindle past 2020, or are held by only a few suppliers, the ability to supply 'growth from a fertiliser bag' will start to affect local supplies and world trade. Vaclav Smil's story of innovation in humankind's modification of the global nitrogen cycle suggests that without industrial nitrogen fixation, global food production could only support a maximum population of three billion under a semi-organic agriculture (Smil 2004).

[15] Conway and Toenniessen 1999; Mann 1999; Tilman *et al.* 2002. Compare also with Section 4.3 in this work on Structural Decomposition Analysis.

2.2 Land use, biodiversity and ecosystem functioning

2.2.1 Insights from Ecology

Ecosystems provide a range of goods (ecosystem properties that have direct market values, e.g food, construction materials) and services (e.g. maintaining hydrological cycles, regulating climate, storing and cycling of nutrients) that directly or indirectly benefit humans (Christenesen *et al.* 1996, Daily 1997). Understanding the influence of land use change and changes to biodiversity on the ability of ecosystems to continue to provide these resources is critical to assessing the sustainability of human activities. Land use change is predicted to have the largest global impact on biodiversity by the year 2100 (Sala *et al.* 2000) and land transformation to yield goods is noted as the most substantial human alteration of Earth's ecosystems (Vitousek *et al.* 1997). Declines in biodiversity are already evident in many areas particularly where natural systems have been converted to croplands, timber plantations, aquaculture and other managed ecosystems (Naeem *et al.* 1999). Three primary reasons to maintain high biodiversity include defending against risk from disturbance, extending the use of resources and providing multiple goods and services (Hooper *et al.* 2005). Increasing species richness can also decrease temporal variability in ecosystem processes (McGrady-Steed *et al.* 1997; Naeem and Li 1997; Emmerson *et al.* 2001) which is important for buffering the effects of disturbance, and can ensure that resource provision is continual. The ability to resist disturbance is especially critical given the current climate of increased land use change.

The relationship between biodiversity and ecosystem function is not clear cut and over 50 patterns have been proposed (Loreau 1998; Naeem 2002). Establishing the link between biodiversity and ecosystem functioning is difficult as the loss of species is usually accompanied by disturbance such as habitat conversion that directly affects many ecological processes and can disguise the impacts of species loss on functioning (Naeem *et al.* 1999; Hooper *et al.* 2005). The idea that loss in plant species is detrimental to ecosystem functioning remains contentious (Aarssen 1997; Grime 1997; Huston 1997). However, evidence from experimental and observational studies suggest that a large suite of species is required to maintain ecosystem function in areas increasingly subjected to human impacts (Loreau *et al.* 2001). Reductions in regional and local diversity can lead to declines in plant production, reduced ecosystem resistance and increasing variability in ecosystem processes such as soil nitrogen levels, water use and pest and disease cycles (Naeem *et al.* 1999).

It is difficult to quantify the link between biodiversity and ecosystem function because the role of species in ecosystem functioning encompasses their diversity, evenness and identity (Naeem *et al.* 1999, Purvis and Hector 2000, Aarssen 2001), not solely their number. The nature of the relationship will also depend on the degree of dominance by the species lost, the strength of interactions with other species and the functional traits of species lost and those remaining (Vitousek and Hooper 1993; Lawton 1994; Naeem 1998). Functional diversity is a major determinant of ecosystem properties (Chapin *et al.* 1997; Chapin *et al.* 2000) and knowing which species or functional types are present is at least as important as knowing how many are present (Aarssen 2001, Hooper *et al.* 2005). Functional diversity increases the stability of ecosystem properties as a suite of species that use resources differently will be able to respond to a range of disturbance types (Hooper *et al.* 2005). For example, MacGillivray and Grime 1995 showed that differences in the responses of five ecosystems to frost, drought, and burning were predictable from the functional traits of the dominant plants but were independent of plant diversity.

Secondly, not all species are created equal and some species, known as keystone species, can have a disproportionate effect on ecosystem function, relative to their abundance (Power *et al.* 1996). The influence of a single species on the function of an ecosystem has been clearly illustrated by extensive research on invasive organisms. Species or functional groups can have a large influence on ecosystem properties (Levine *et al.* 2003). Similarly the loss of keystone species can dramatically alter ecosystem function (Browne and Heske 1990). Costs incurred by society are difficult to predict when the relationship between biodiversity and ecosystem functioning is non-linear. The loss of keystone species can result in sudden declines in function due to a threshold effect (Chapin *et al.* 2000).

2.2.2 Insights from Life-Cycle Assessment

Over the past 10 years there has been extensive debate within the Life Cycle Analysis (LCA) community on how best to quantify land use impacts.[16] While some impacts of land use are already consistently treated within LCA, including climate change, eutrophication and acidification, and toxicity, consensus has yet to be reached on the best way of incorporating impacts on more direct aspects of ecosystem functioning, such as biodiversity and soil quality (Jolliet and Müller-Wenk 2004; Udo de Haes and Heijungs 2004; Milà i Canals *et al.* 2007).

The Land Use taskforce within the UNEP-SETAC Life Cycle Initiative (http://www.uneptie.org/pc/sustain/lcinitiative/) recently recommended that characterisation factors for land use impacts should embody all the following information: impacts on biodiversity and soil quality; impacts from climate change, eutrophication and acidification, toxicity etc.; whether the impact is from transformation or occupation; the size of the impact relative to a suitable reference state (depends on coverage and intensity of land use, and the biogeographical conditions of the area); and the time dependence of land use on impact.[17]

Given the lack of such detailed data on a global scale, it is not currently possible to achieve this ideal. However, a number of simplified weighting schemes for land use impacts have been suggested over the years as a first approximation. What follows is a brief summary of methods for which some characterisation factors have been published. Discussion of other methods can be found in the references cited above.

Lindeijer 2000a provides an extensive review of methods for weighting land use impacts and tabulates values for 5 proposed systems. Three of these come from Knoepfel 1995 and are based on: the degree of biological accumulation (Whittaker and Likens 1973) defined as gross primary production; natural regeneration time (Hampicke 1991); and people's perception of the value of each land use class (Jarass *et al.* 1989; Bradley and Peter 1998). The two other systems are based on multiple criteria. Auhagen 1994 use a combination of naturalness, species diversity, rareness of biotopes, and density of individual plants and animals, while Felten and Glod 1995 reduce this to two criteria, the Simpson index for

[16] Blonk *et al.* 1997; Heijungs *et al.* 1997; Baitz *et al.* 1998; Ekvall 1998; Müller-Wenk 1998; Swan 1998; van Dobben *et al.* 1998; Köllner 2000; Lindeijer 2000a; b; Mattsson *et al.* 2000; Goedkoop and Spriemsma 2001; Schenck 2001; Weidema and Lindeijer 2001; Brentrup and Küsters 2002; Udo de Haes and Heijungs 2004; Geneletti 2007; Milà i Canals *et al.* 2007.

[17] See Milà i Canals *et al.* 2007, and a critique by Udo de Haes 2006. Udo de Haes and Heijungs 2004 cite strong site dependence as being one of the main challenges in incorporating land use into LCA, in particular when changing land use type.

biodiversity (Simpson 1949) and an indicator based on the IUCN Red List for endangered species.

Swan and Pettersson 1998 use bioproductivity and biodiversity measures to characterise the ‘actual state’ ($1 - p$) of land, while van Dobben *et al.* 1998 and Köllner 2000 approximate ecosystem quality by the species diversity of vascular plants, because they are most readily able to be surveyed, and provide habitat and food to other species. Based on species-area relationships, the latter two authors define the ‘species richness’ (α) and the ‘species-pool effect potential’ (SPEP) as a function of the number of species on a particular land type and the average number of species in the surrounding reference region.

While a considerable numbers of methods for calculating characterisation factors have been proposed, most of them still suffer from a lack of data to make them operational on a global scale. As a possible solution to this, Lindeijer 2000b has suggested using a combination of two indicators, the density of vascular plant species and fNPP.[18] For both indicators, reference values were generated on a global scale based on a literature review of the most recent (at the time) scientific measurements. A few sample impact factors are given for specific mining and logging operations, however it is noted that more data needs to be gathered for a more general use of the method. Finally, it is suggested that these indicators be complemented with a water resource indicator and a climate change indicator to capture the full impact of land use.

In order to give a more complete indication of biodiversity impacts, Cowell 1998 has proposed combining the diversity of species with three additional indicators - the relative area of a specific ecosystem type, the number of rare species, and the NPP[18] as an indicator for the number of individuals – but provides no actual characterisation factors.

Along similar lines, Weidema and Lindeijer 2001 propose using two indicators: NPP for ecosystem productivity and life support, and a composite variable for biodiversity based on species richness, ecosystem scarcity and ecosystem vulnerability. They also provide global reference values for NPP and the diversity of vascular plant species, together with global averages for occupation impact on these indicators for a selection of land uses.

More recently, Wagendorp *et al.* 2006 have suggested some novel indicators based on the theory that ecosystems maximise their dissipation of external exergy[19] fluxes through maximizing their internal exergy storage in the form of biomass, biodiversity and complex trophical networks. It is suggested that human impacts will decrease this ecosystem exergy level via simplification, e.g. by decreasing biomass and destroying the internal complexity of the ecosystem. One of the indicators investigated is the thermal response number (TRN), which can be computed from thermal remote sensing data and radiation measurements, and so at least in theory is available on a global scale.

2.2.3 Insights from cross-country analyses

A number of studies have emerged that seek to identify threats from a global perspective, by examining country-level data and searching for statistical determinants of the number of

[18] Free net primary biomass production (fNPP) is the amount of biomass which nature can apply for its own development, i.e. net produced biomass (NPP) minus human consumption of biomass.
[19] Defined as energy able to do work, i.e. subtracted of its entropic content.

threatened species. One reason for global assessments is to find drivers of species threats beyond the plethora of locally specific circumstances, in order to inform policy-making at the national and international levels.

Population density and per-capita Gross Domestic Product (GDP) are frequently amongst the determinants tested, with many authors searching for an inverted-U relationship, or an "environmental Kuznets curve" (Naidoo and Adamowicz 2001; Asafu-Adjaye 2003; McPherson and Nieswiadomy 2005; Pandit and Laband 2007a; b).[20] The results of these studies show a high variability, and a generally weak connection between explanatory and explained variables. In most of these studies, typical significant correlation coefficients range around ±0.3 to ±0.5, and typical regression R^2 are around 0.5. None of the studies resembles another in their choice of functional specification, set of countries, or selection of explanatory variables, so that direct comparisons are not possible.

The work by Pandit and Laband 2007a; b and McPherson and Nieswiadomy 2005 is particularly interesting in that it is one of the few that demonstrate the importance of considering that factors influencing species imperilment in one country may also influence species imperilment in neighbouring countries. Both teams exploit spatial autocorrelation[21] in order to incorporate cross-border effects into their regression models. In addition, Pandit and Laband 2007b experiment with differently weighted adjacency matrices in order to uncover specific reasons for cross-border spillover threats.

Instead of socio-economic and demographic variables, Lenzen *et al.* 2007b use land use information as potential determinants of species threats, because land use is a closer proxy to causes for biodiversity loss than for example population density or GDP.[22] They argue that for example in the case of Australia, even though population density is extremely low especially in rural areas, the latter are used intensively for cropping and grazing for exports, leading to significant biodiversity loss (Glanznig 1995). Similarly, wealthy countries such as Switzerland may preserve their own biodiversity relatively well through displacing any environmentally damaging domestic production by shifting to or importing from other countries (Muradian *et al.* 2002). In explaining the number of threatened species in one country, Lenzen *et al.* 2007b explicitly use information on land use patterns in all neighbouring countries, and on the extent of the country's sea border. Their results show that cross-border land use patterns have a significant influence on species threats, and that land use patterns explain threatened species better than less proximate socio-economic variables.

[20] Proponents of the environmental Kuznets hypothesis argue that with increasing development and wealth, environmental pressure and impact initially increase, but then decrease again once societies have attained a level of prosperity that allows them to improve environmental conditions. For further reading see Pearson 1994; Selden and Song 1994; Shafik 1994; Grossman and Krueger 1995; Stern *et al.* 1996; Cole *et al.* 1997; Ekins 1997; Ehrhardt-Martinez *et al.* 2002; Stern 2003.

[21] Spatial autocorrelation can be modelled either in a spatial lag model where the explained variable appears also as a (spatially weighted) dependent variable, together with other determinants, or in a spatial error model, where the error term assumes a particular spatially autoregressive shape.

[22] This agrees with the assessment of Kerr and Currie 1995, that "it seems unlikely that […] socioeconomic variables are the proximate influences on species extinction. Rather, species survival is probably most often endangered by specific human activities such as habitat modification, hunting, and so forth". Similarly, Asafu-Adjaye 2003 indicates that a variable such as the percentage of GDP in agriculture is only a proxy for habitat conversion, which in turn is one of the major threats to biodiversity. Land use intensity and land use changes are the most frequently listed direct driving force for biodiversity loss in Spangenberg 2007, and in a Special Issue of *Basic and Applied Ecology* (Poschlod *et al.* 2005). Socio-economic and demographic variable have been shown to have an influence on land use patterns (Deacon 1994), which in turn affect species. Eppink *et al.* 2004 and Mattison and Norris 2005 argue for an integration of land use economics and biodiversity ecology.

Study	Lenzen *et al.* 2007b	Lenzen *et al.* 2007b	Lenzen *et al.* 2007b	Swan and Pettersson 1998	van Dobben *et al.* 1998	Weidema and Lindeijer 2001		Reidsma *et al.* 2006
Spatial level	World	World	World	Not defined	The Netherlands	World	World	European Union
Variable	Percentage of threatened birds	Percentage of threatened mammals	Percentage of threatened plants	Bioproductivity and biodiversity, $1-p$	Species diversity of vascular plants, $1-\alpha$	NPP [h]	Biodiversity [i]	Ecosystem quality
Method	Multiple regression [a]	Multiple regression [a]	Multiple regression [a]	Guesstimates	Expert judgement [e]	Estimates derived from published data		Literature review [j]
Built-up land	-0.46 ± 0.18	0.05 ± 0.34	0.44 ± 0.14	0.99	1.0	0.86	1.00	
Permanent crops	0.51 ± 0.07	0.67 ± 0.14	0.18 ± 0.03	0.76 [b]	0.6 [f]	0.37	1.00	0.8 ± 0.15 [k]
Permanent pasture	0.09 ± 0.01	0.09 ± 0.02	0.01 ± 0.01			0.14	0.34	0.73 ± 0.13 [k]
Non-permanent arable	0.00 ± 0.03	0.08 ± 0.05	-0.04 ± 0.02	0.2 [c] – 0.34 [d]				
Timber plantations	0.45 ± 0.07	0.72 ± 0.13	-0.13 ± 0.07	0.85	0.2			
Natural forest	0.03 ± 0.01	0.11 ± 0.02	0.00 ± 0.01	0.04	0.0 [g]			
Non-arable land	0.02 ± 0.01	0.12 ± 0.02	0.05 ± 0.01					0.00 [l]

Tab. 1: Summary of characterisation factors of land use documented in the literature. [a] Weighted Last Squares, linear specification, standard deviation of regression coefficients obtained using *t* test; [b] "normal agriculture"; [c] "sustained forestry", [d] "productivity forestry; [e] based on number of vascular plant species per km^2 in the Netherlands (Witte and van der Meijden 1995); [f] "intensive agriculture"; [g] "pristine"; [h] NPP values for broadly categorized terrestrial ecosystems (Amthor and Members of the Ecosystems Working Group 1998); [i] existing (USGS[23]) and potential (Prentice *et al.* 1992) biome areas, and species richness per biome (Barthlott *et al.* 1996); [j] to determine the dose-effect relationship of the intensity of agricultural land use on biodiversity; [k] full ranges; [l] "natural grassland".

[23] Sourced from the USGS website http://edcwww.cr.usgs.gov/eros-home.html.

Study	Köllner 2000		Knoepfel 1995		Auhagen 1994	Felten and Glod 1995
Spatial level	Germany and Switzerland [m]	World	Unknown [s]	Unknown [s]	Unknown [s]	Unknown [s]
Variable	Species diversity of vascular plants, ΔSPEP [n]	Biological accumulation	Regeneration time	Private perception of land value	Naturalness, species diversity, rareness of biotopes, density of plants and animals	Biodiversity, endangered species
Method	Linear regression	Based on Whittaker and Likens 1973	Based on Hampicke 1991	Based on Jarass *et al.* 1989 & Bradley and Peter 1998	Unknown [s]	Unknown [s]
Built-up land	1.0	0.95	0.9996	0.71	0.94	0.85
Permanent crops	0.55	0.90 [p]	0.9953 [p]	0.48 [p]	0.82 [p]	0.51 [p]
Permanent pasture	0.51					
Non-permanent arable	0.33 [o]	0.00 [q]	0.83 [q]	0.16 [q]	0.65 [q]	0.15 [q]
Timber plantations						
Natural forest	0.00	0.00 [r]	0.00 [r]	0.00 [r]	0.00 [r]	0.00 [r]
Non-arable land						

Tab. 1 (cont.): Summary of characterisation factors of land use documented in the literature; [m] applied to Spain by Antón *et al.* 2007; [n] normalised to built-up = 1; [o] “less intensive meadow”; [p] “cultivated system (intensive)”; [q] “cultivated system (extensive)”; [r] “natural systems”; [s] unable to obtain the original document.

2.2.4 Quantitative factors linking land use and biodiversity

Land use characterisation factors for a range of different LCA and non-LCA methods are summarised and compared in Table 1. Note that different authors use different classification schemes for land types. Here we have used a reduced set of 7 land types and used personal judgment to determine their equivalences to other classification schemes in the literature. Ranges have been given where more than one land type from the literature was judged to lie within one of our specified land types. We also attempted to find compromise values for factors obtained from those studies dealing with biodiversity variables (Tab. 2). These compromise values will be used in our temporal analysis in Section 4.

Land use type	Characterisation factor	Standard deviation
Built-up land	0.86	± 0.19
Permanent crops	0.62	± 0.23
Permanent pasture	0.32	± 0.28
Timber plantations	0.42	± 0.40
Non-permanent arable	0.47	± 0.22

Tab. 2: Averages and standard deviations for a subset of characterisation factors from Tab. 1, dealing with biodiversity variables (all studies except Weidema and Lindeijer 2001, Knoepfel 1995, and Reidsma *et al.* 2006).

The purpose of land use characterisation factors is to enable land-use-specific estimates of biodiversity impacts, where species data at the land-use-type level does not exist. Applying these factors therefore involves a trade-off between a) a potentially poor proxy (land use) based on good data, and b) a direct representation of species diversity based on underreported, or otherwise poor data (compare with Voigtländer *et al.* 2004).

2.3 Human consumption and greenhouse gas emissions

The amount of greenhouse gases emitted by humans depends on factors that are similar to the land use analysis in Section 2.1. Here, these are, amongst others, population, living standards (affluence), energy consumption, greenhouse gas contents of fuels, the number of livestock units, the area of land cleared, etc. For example, CO_2 emissions G_e from energy use can be written as a function of population P, gross economic output per capita (x, \$PPP/cap), energy intensity (e, Megajoule/\$PPP), and greenhouse gas content (g, tonnes/Megajoule) as $G_e = P\,x\,e\,g$. Similarly, CH_4 and N_2O emissions are proportional to agricultural output. Common drivers for all types of emissions are population and gross economic output; these have been dealt with in Section 2.1.

2.3.1 Fuel combustion

The most important component of global greenhouse gas emissions is CO_2 from the combustion of fossil fuels. As with material intensities, energy intensities can be analysed using the concept of elasticity (see Section 2.1). Lenzen *et al.* 2006 pool and regress cross-country data on per-capita energy intensities *e*, and find $\eta(x) = 0.963 - 0.022 \ln(x)$, where x is in units of '000$PPP/cap ('Regr' and 'eta' in Fig. 8). [24,25] Their results are in broad agreement with 1997 data reported by Ang and Liu 2006.

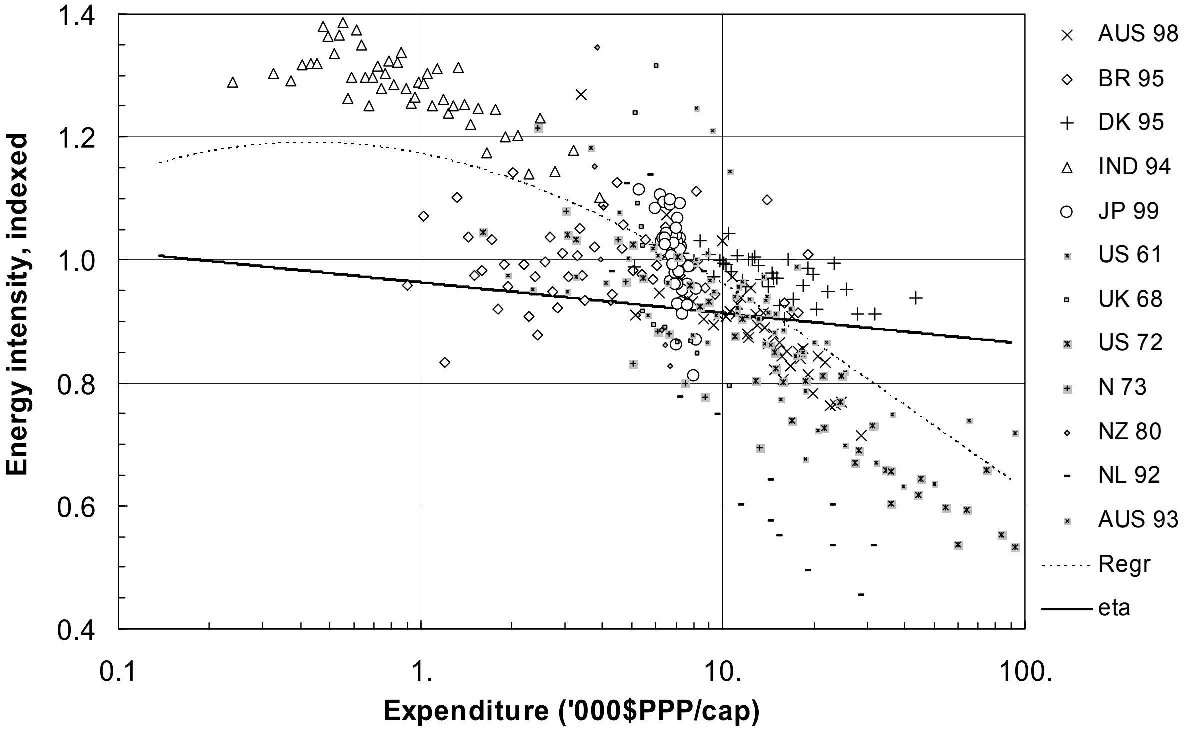

Fig. 8: Comparison of energy intensities as a function of household expenditure ('000$PPP/cap). Values are indexed to the average energy intensity of all observations between 6,000$PPP/cap and 9,000$PPP/cap. Results from Lenzen *et al.* 2006 (large symbols): Australia (AUS 98), Brazil (BR 95), Denmark (DK 95), India (IND 94), and Japan (JP 99). Results from other studies (grey small symbols): US 1961 (Herendeen and Tanaka 1976), UK 1968 (Roberts 1975, p. 766), US 1972 (Herendeen *et al.* 1981), Norway 1973 (Herendeen 1978), New Zealand 1980 (Peet *et al.* 1985), Netherlands 1992 (Vringer and Blok 1995), Australia 1993 (Lenzen 1998). 'Regr' denotes a regression of all data in the form $e = e^{(y=1)} y^{\eta(y)}$, with 'eta' being the elasticity $\eta(y)$.

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[24] The World Bank (http://www.worldbank.org/depweb/english/modules/glossary.htm#ppp) defines PPP as "a method of measuring the relative purchasing power of different countries' currencies over the same types of goods and services. Because goods and services may cost more in one country than in another, PPP allows us to make more accurate comparisons of standards of living across countries."

[25] Lenzen *et al.* 2006 use double-log specifications in order to impose correct asymptotic restrictions, that is to avoid negative values for energy consumption (de Bruyn 2000, p. 94; Stern 2003, p. 6). An expenditure elasticity of the form $\eta(y) = \eta_0 + \eta_1 \ln(y)$ ("eta" in Fig. 8) then results from a regression according to $\ln(e) = k + \eta_0 \ln(y) + \eta_1 \ln(y)^2 + \Sigma_i F_i$, where y is per-capita expenditure. Fixed country effects are taken care of by the inclusion of dummy variables F_i, where $F_i = 1$ for country i, and 0 otherwise. Final consumption expenditure y is roughly proportional to gross economic output x, so that roughly the same elasticity should apply.

Lenzen *et al.* 2006 stress that – strictly speaking – their data does not support the existence of a single, uniform cross-country relationship between energy requirements and Gross National Expenditure: Instead, elasticities vary across countries, even after controlling for socioeconomic-demographic variables. This result confirms previous findings in that trends are unique to each country, and determined by distinctive features such as resource endowment, historical events, socio-cultural norms or political system.

In contrast to agricultural output there are economies of scale for energy use, because – like food – people are able to share energy, for example in form of electricity for appliances and lighting. Accordingly, in their study of five countries, Lenzen *et al.* 2006 found a significant albeit weak population effect.

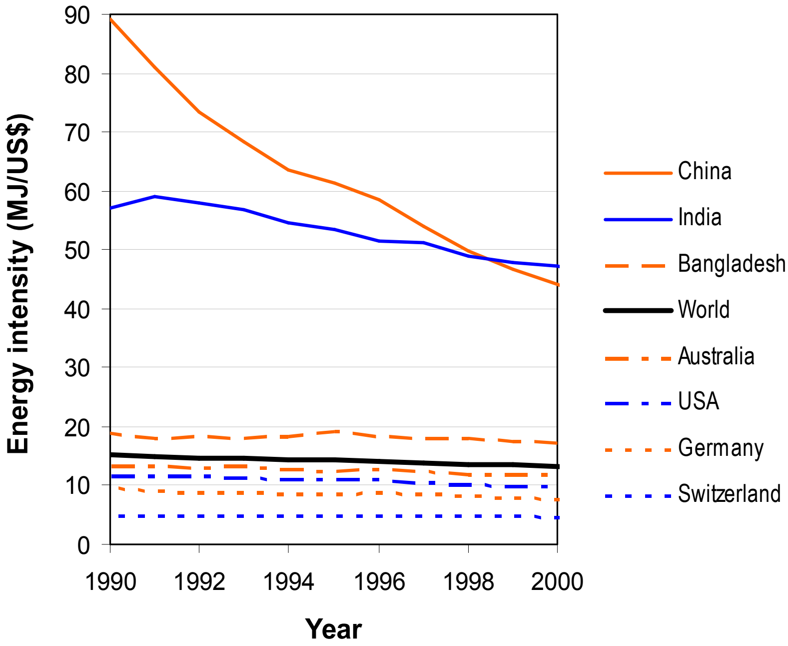

Fig. 9: Energy intensity trends of several countries (primary energy MJ per 2000$US; World Resources Institute 2006).

At the country level, conversion efficiencies and carbon contents of energy technologies vary greatly. The most important factors are availability of energy resources, combusted fuel type and quality, the mix of electricity conversion technologies (for example coal-fired versus nuclear power), age and sophistication of electricity infrastructure, geography and electricity grid losses, extent of the utilisation of waste heat, end-use efficiency of energy use, climate, transport infrastructure, and vehicle size and efficiency. Energy intensities (e) are strongly related to level of development, as shown in the decadal trend in Fig. 9. Rapidly developing countries such as China and to a lesser extent India, show extreme falls in energy intensity in the main expansionary phase of economic development. Despite these trends, energy intensities of these countries can still be an order of magnitude higher than those for the most developed service economies, such as Switzerland. Carbon intensities follow similar trends as for energy intensity, but with differences due primarily to the overall greenhouse gas content of the energy mix (Fig. 10). For example, China and India have significant coal reserves and substantial coal-fired electricity generation capacity (Graus *et al.* 2007).

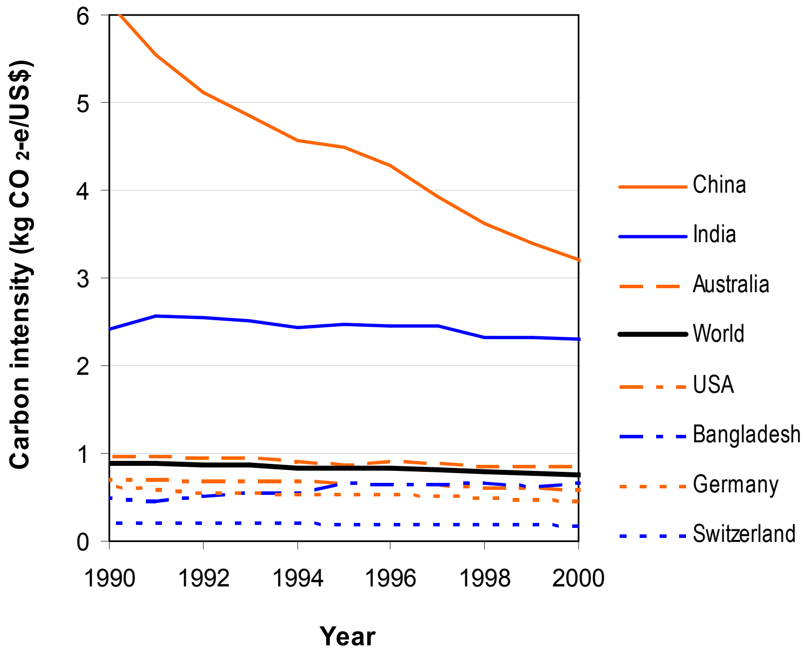

Fig. 10: Carbon intensity trends of several countries (energy-related carbon dioxide emissions per 2000$US; World Resources Institute 2006).

For the purpose of projecting energy-related greenhouse emissions, the most important trend is the greenhouse gas content *g* (the combination of data in Figs. 9 and 10, shown in Fig 11), which is influenced by a range of factors specific to a country's stage of economic development (compare Mielnik and Goldemberg 1999 with Ang 1999).[26] The strongest driver of absolute value of *g* is the local availability of energy sources, particularly coal compared with natural gas for electricity generation. Changes in energy quality, primarily an increase in the proportion of electricity (and its generation efficiency) compared with thermal and transport energy use, also affect *g* (Graus *et al.* 2007). Further, since carbon contents are strongly dependent on electricity infrastructure and a country's energy resource base, both with generally long time frames, changes in carbon content are less dramatic than energy and carbon intensity changes.

[26] The apparent contradiction of these views can perhaps be explained by the "three laws of energy transitions" by Bashmakov 2007: "the law of stable long-term energy costs to income ratio; the law of improving energy quality; and the law of growing energy productivity".

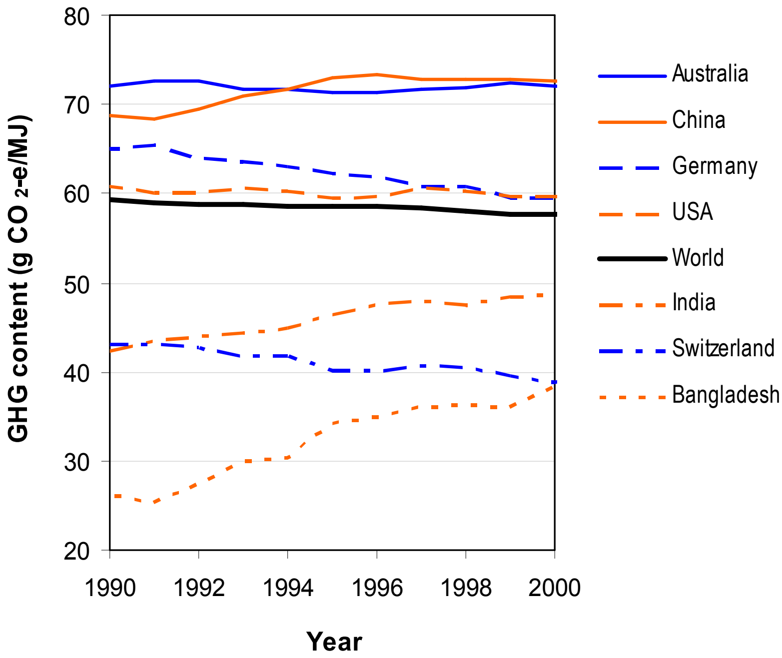

Fig. 11: Greenhouse gas content trends of several countries (energy-related carbon dioxide emissions per primary energy use; World Resources Institute 2006).

Plotted against per capita figures for 2002 (Fig. 12), the carbon content reveals a classical inverted-⋃ Kuznets form[20]. In this view, the decarbonisation of energy use is a consequence of increasing wealth, bringing about increasing use of renewable and low-carbon sources. As with Kuznets' initial hypothesis about income equality (Kuznets 1955), carbon contents are set to increase during the initial stages of development, but eventually decline as countries have completed their transitions to affluent societies.

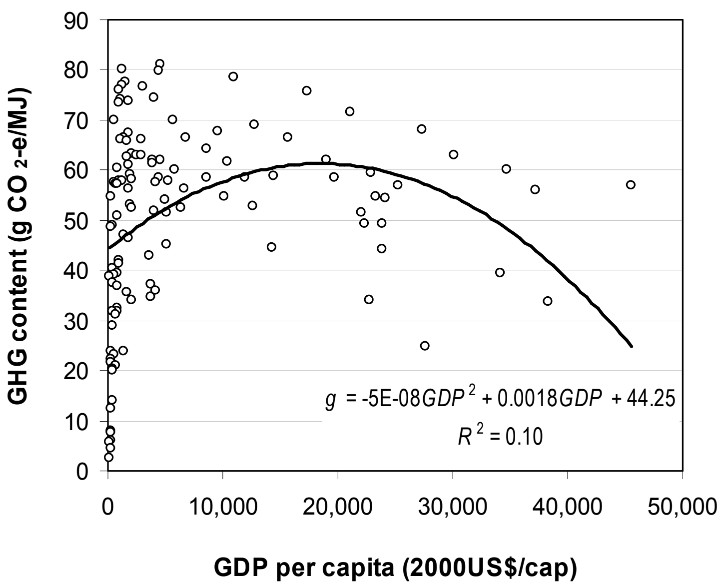

Fig. 12: Per capita greenhouse gas content in 2002 (energy-related carbon dioxide emissions per 2000$US) by country (World Resources Institute 2006).

2.3.2 Agriculture, forestry, and forest conversion

Enteric fermentation in livestock and animal manure are major global sources of CH_4 and N_2O. In their reporting guidelines, the Task Force on National Greenhouse Gas Inventories 1996 of the Intergovernmental Panel on Climate Change (IPCC) provides a calculation recipe for national emissions from these sources, which is based on the number of livestock, and region- and climate-specific emission factors. In this work we attempt to forecast livestock numbers based on population and demand, and convert to greenhouse gases following the IPCC procedures. Accordingly, we distinguish beef cattle, dairy cattle, sheep, goats, pigs and poultry, and seven different animal waste management systems.

Emissions from land use changes occur when land is cleared for agriculture: the removed biomass is partly immediately burned or partly decays over many years, and the disturbed soil emanates CO_2 over 20 years or so. These sources are possibly the least understood and most uncertain of all. In this work we attempt to model only forest and grassland conversion, which causes CO_2 emissions during burning of above-ground biomass, delayed decay of above-ground biomass over 10 years, and delayed below-ground exponential releases of soil carbon over 20 years. Today, the main clearing activities occur in tropical areas.

Finally, plantations and managed forests take up CO_2 through harvesting of timber for non-fuel uses. Here we estimate CO_2 sequestration of commercial and non-commercial forests based on annual volumetric increments, a carbon fraction of 50% dry matter, and conversion rates of between 0.9 and 1.0 tonnes dry matter per cubic metre commercial roundwood (Task Force on National Greenhouse Gas Inventories 1996).

2.4 Greenhouse gas emissions and temperature increase

The emission of greenhouse gases causes global warming through building up atmospheric concentrations, and resulting increase in radiative forcing, which in turn has a delayed effect on global temperature rise. This is a complex causal chain that involves many non-linear and/or feedback effects, which Global Circulation Models aim to capture. A workable Ecological Footprint method would benefit not only from a simplification of these models, but also from a formulation that could isolate the effect of a particular "parcel" of greenhouse gas emissions at a particular point in time on future temperature changes. This is exactly what the authors of the Brazilian proposal to the IPCC (Federative Republic of Brazil 1997) had in mind when they argued that the responsibility of countries for climate change should not be measured as contributions to greenhouse gas emissions, but as contributions to global warming. Measured as such, developed countries would have to shoulder additional responsibility for historical emissions, because these greenhouse gases have resided in the atmosphere for much longer than more recent emissions from developing countries, thus causing more radiative forcing.

In order to be able to quantify the contributions of single countries to global warming, the effect of emitting entities on temperature changes has to be approximated by formulating a separable emissions-temperature relationship (Meira and Miguez 2000) by imposing addition invariance as $\Delta T(G_1 + G_2, t) = \Delta T(G_1, t) + \Delta T(G_2, t)$. While this approximation ignores non-linearities, the authors argue that it represents a fairer "policy-maker model" than the conventional Global Warming Potentials, because it adequately reflects temporal effects (see

Rosa and Schaeffer 1993; 1995). Following the separability condition above leads to a simplified mathematical expression linking greenhouse gas emissions and temperature increase (see Section 3.1).

2.5 Temperature increase and bioproductivity

Future climate change will directly affect the productivity of both natural and agricultural ecosystems (path 13). There are many factors involved leading to changes in different directions, depending on the region (IPCC Working Group II 2007). Increased agricultural yields can be expected through warmer days and nights in colder environments, whereas yields and bioproductivity in general are likely or very likely to decrease through heat waves, droughts, fire, soil erosion, flooding, extreme weather events or salinisation.

Changes in temperature and precipitation regimes as well as direct effects of greenhouse gas emissions through CO_2 fertilisation are recognised as the major drivers of changes in ecosystem productivity. However, their relative role and strength are still debated and unclear, in particular when vegetation-climate feedbacks are considered (Fung *et al.* 2005). Data on observed past and modeled future changes in productivity indicate that both direction and size of this effect are likely to vary greatly between regions: regions not limited by water availability have experienced an increase of forest productivity in the past (Boisvenue and Running 2006) and are likely to experience an increase in net primary productivity (NPP) with increasing temperatures in the future (Thuiller *et al.* 2006; Morales *et al.* 2007) in regions limited by water availability, productivity is likely to decrease, potentially making ecosystems in these regions carbon sources (Fung *et al.* 2005; Morales *et al.* 2007).

Despite regional differences, net biome productivity is likely to increase at the global scale (Gitay *et al.* 2002). Global ecosystem net primary productivity is predicted to increase by ca. 40% (average of six global vegetation models) between 2000 and 2100 for a projected temperature increase of about 3.9ºC over the same period (Cramer *et al.* 2001). Using ten regional climate models, Morales *et al.* 2007 predict an average increase in NPP of ca. 18 % for a projected temperature increase of ca. 4.6ºC in Europe's ecosystems between the 1961-1990 and the 2071-2100 period. Ewert *et al.* 2005 estimate a 30% increase in crop yields if present CO_2 concentrations were doubled. These ecosystem-wide, large-scale studies use models of the potential distribution of natural ecosystems and biomes to assess future changes in productivity and their results are therefore only of limited use for estimates of change in productivity in agricultural systems.

It is valid to assume that if NPP is projected to increase for all vegetation types, it is also likely to increase for crops. Agricultural yield has indeed steadily increased over the last 50 years or so but it is likely that not only climate change and CO_2 fertilisation but also socioeconomic as well as agricultural and other land use factors have caused this trend (Long *et al.* 2006; Ewert *et al.*; Long *et al.* 2007). The Fourth Assessment Report of the IPCC (IPCC Working Group II 2007) concludes that globally, the potential for food production is projected to increase with increases in local average temperature over a range of 1-3°C. Commercial timber productivity is also projected to rise modestly with climate change in the short- to medium-term, with large regional variability around the global trend. However, the assessment report also concludes that above 3°C production is projected to decrease. At lower latitudes, especially seasonally dry and tropical regions, crop productivity is projected to decrease for even small local temperature increases (1-2°C) due to heat and drought.[27]

A multiple regression analysis of the data provided by (Morales *et al.* 2007) suggests that a 1ºC temperature increase alone accounts for a 3% increase in productivity. The very comprehensive study on global estimates for projected (2000 - 2100) NPP using six different vegetation models (Cramer *et al.* 2001) suggests an even higher increase of about 10% in NPP per 1ºC temperature rise. In Section 3, we use a compromise of the two studies as a working equation in this work and define a linear relationship of the $\mathrm{d}p/p$ – $\mathrm{d}T/T$ coefficient to decrease from 6% at 0.5ºC temperature anomaly to 0 at 3ºC temperature anomaly.

2.6 Temperature increase and biodiversity

Climate is the ultimate driver of species distributions and species diversity patterns at large spatial scales. Species persist in climatic conditions in which they can establish viable populations given other constraining factors such as nutrient availability, disturbance regimes and biotic interactions. The geographic distributions of species, species assemblages and species diversity has shifted under changing climates in the past. There is already ample evidence (Hickling *et al.* 2006; Menzel *et al.* 2006) for recent range shifts and phenological changes in a large number of taxonomic groups and regions. These changes in species occurrence and performance will ultimately lead not only to changes in biodiversity but also to functional shifts and related changes to ecosystem services. Under the valid assumption that climate is one of the main drivers of species distributions, it is clear that future climate change will affect species distributions. Currently recognised biomes and centres of high species diversity are projected to be at risk from land use and climate change (Sala *et al.* 2000; MA 2005).

Current estimates of future extinction rates under climate change often employ the species-area relationship to assess species loss as a function of change in projected climatically suitable area per species directly (Thomas *et al.* 2004; Thuiller *et al.* 2005) or as a function of change in projected biome area (Malcolm *et al.* 2006; van Vuuren *et al.* 2006). The uncertainties and shortcomings associated with these approaches are now widely acknowledged (Pearson and Dawson 2003; Hampe 2004; Guisan and Thuiller 2005; Araújo and Rahbek 2006; Dormann 2007); they include among others: i) disequilibrium of current vegetation and climate. ii) limited knowledge of individual species behaviour (demography, dispersal, biotic interactions), iii) unknown species response curves. In addition to these uncertainties associated with future projections of species distributions, the relative roles of

[27] In its Fourth Assessment Report, the IPCC Working Group II 2007 writes "Crop productivity is projected to increase slightly at mid to high latitudes for local mean temperature increases of up to 1-3C depending on the crop, and then decrease beyond that in some regions (5.4) … At lower latitudes, especially seasonally dry and tropical regions, crop productivity is projected to decrease for even small local temperature increases (1-2C), which would increase risk of hunger (5.4) ... Globally, the potential for food production is projected to increase with increases in local average temperature over a range of 1-3C, but above this it is projected to decrease (5.4, 5.ES) … Adaptations such as altered cultivars and planting times allow low and mid- to high latitude cereal yields to be maintained at or above baseline yields for modest warming (5.5) … Increases in the frequency of droughts and floods are projected to affect local production negatively, especially in subsistence sectors at low latitudes (5.4, 5.ES) … Globally, commercial timber productivity rises modestly with climate change in the short- to medium-term, with large regional variability around the global trend (5.4) … Regional changes in the distribution and production of particular fish species are expected due to continued warming, with adverse effects projected for aquaculture and fisheries (5.4.6).

past vs. current climate and ecological vs. evolutionary factors in explaining large-scale diversity patterns is still unclear (Kreft and Jetz 2007; Rahbek *et al.* 2007). This further complicates the formulation of current climate-diversity relationships and resulting future projections of diversity change under climate change.

An alternative approach for quantifying levels of risk to biodiversity under climate change is to quantify the magnitude and geographic shifts in climate space and relate these shifts to biodiversity information directly. This approach has the potential to offer biologically meaningful risk assessment of large areas while potentially cutting out some of the sources of uncertainty described above. The rationale underlying this approach is the assumption that species occurring at a location are adapted to the climatic conditions of that location. If these conditions change in the future, the species will either i) have to adapt to the new conditions, ii) disperse and migrate in order to find other areas with climate conditions analogous to those they are used to, or iii) go extinct from that location. Under this reasoning, the risk to biodiversity at a location is high if the future climate conditions are far outside their current bounds and if there is no or only small areas with analogous climate conditions to which species could potentially migrate too. These measures of climate risk have recently been quantified for Europe (Ohlemüller *et al.* 2006) and globally (Williams *et al.* 2007).

Here we suggest a way of how such climate risk surfaces might potentially be used to quantify future risk to biodiversity. Based on Williams *et al.* 2007 (Fig. 4B, p. 5741), we derive a generic global relationship between projected changes in global temperature and the percentage global land area with disappearing climates by 2100. Following well established species-area approaches, we then translate losses in climatically analogous area into expected losses in species richness. Assuming that 15% of the global terrestrial area with disappearing climates under a 2ºC temperature increase equates to a threat to 15% of the earth's species we infer a global temperature-species-threat relationship as $S = (0.268 * T) - 0.137$, that is a 1 ºC temperature increase will lead to 13.1% of the world's land area to have a climate that is outside current conditions, thereby potentially threatening 13.1% of the world's species. This is a bold assumption given the uneven distribution of both climate disappearance and biodiversity: Most of the climate disappearance seems to occur in biodiversity hotspots (see maps in Williams et al. 2007). Nevertheless, our estimate is also in line with the IPCC Fourth Assessment Report on impacts of climate change (IPCC Working Group II 2007) that concludes that approximately 20-30% of plant and animal species assessed so far are likely to be at increased risk of extinction if increases in global average temperature exceed 1.5-2.5ºC.

Our approach makes a number of assumptions which should be kept in mind when interpreting the results. Firstly it assumes that all change is bad for species living at a location. In reality, some species may actually benefit from changing climate conditions at locations where they currently are. Secondly, as applied here, we assume uniform changes in disappearing climates across the earth's land masses. This is clearly not the case (Williams *et al.* 2007) and estimates should preferably be calculated at grid cell or country level. Keeping this in mind, we believe that our approach can give some indication as to how much and which areas on Earth will face high levels of risks from shifts in climate space and associated risks for biodiversity.

2.7 Biodiversity and bioproductivity

There is strong evidence from experimental studies that increases in plant diversity lead to greater productivity (path 8/9, Naeem *et al.* 1994; Naeem *et al.* 1995; Tilman *et al.* 1996; Tilman *et al.* 1997a; Tilman *et al.* 2001b). One of the better known references supporting this view is probably Naeem *et al.* 1999, who conclude that "plant production may decline as regional and local diversity declines; ecosystem resistance to environmental perturbations, such as drought, may be lessened as biodiversity is reduced; ecosystem processes such as soil nitrogen levels, water use, plant bioproductivity, and pest and disease cycles become more variable as diversity declines". These authors also confirm that "biodiversity declines are already pronounced in many areas, especially where natural ecosystems have been converted to croplands, timber plantations, aquaculture and other managed ecosystems". With regard to distinguishing paths 8 and 9 in Fig. 1 they write: "Determining whether biodiversity per se is important to ecosystem functioning has been difficult, partly because many of the factors such as habitat conversion that reduce local biodiversity also directly affect many ecological processes, masking the more subtle impacts of species loss on functioning". Separating the biodiversity effect out from other effects requires monitoring species diversity and productivity in – generally small-scale – field experiments. Interestingly, this reference is also one of the more heavily criticized (Naeem 2000; Tilman 2000; Wardle *et al.* 2000).

Hector *et al.* 1999 found significant reductions in above ground plant biomass with reductions in plant species and functional diversity in a grassland experiment. The authors found evidence for a linear relationship between productivity and the natural logarithm of plant species, leading to a rule of thumb that halving species diversity leads to a 10% to 20% reduction in productivity (Tilman 1999). Costanza *et al.* 2007 also found evidence for a strong positive relationship between biodiversity and net primary productivity in North American ecosystems, but only at high temperatures (13ºC average). In the high temperature range, biodiversity explained 26% of the variation in net primary productivity and a 10% change in biodiversity corresponded to a 1.73% change in productivity (Costanza *et al.* 2007).

Increases in species or functional richness have also been shown to increase stability in productivity (McNaughton 1977; 1993) and resistance to drought (Tilman and Downing 1994). However, many of these findings have been heavily debated on experimental and statistical grounds (Guterman 2000; Hector *et al.* 2000; Huston *et al.* 2000). A major criticism of such studies is that the method of random species addition or removal typically used do not mimic natural or human induced species loss or addition and therefore have very little biological relevance (Huston *et al.* 2000). Evidence for increased productivity with increased diversity is contrary to observations in nature where the most productive ecosystems are often low in species diversity (Huston 1994, Grime 2001). Variations in productivity at a global scale are driven by variation in resources rather than differences in species diversity (Huston 1994). High plant diversity may in fact be a response to high productivity rather than the cause of it.

Understanding the mechanisms behind observed productivity patterns is at the centre of the debate. Loreau *et al.* 2001 grouped these mechanisms into two main classes. Firstly, processes such as niche differentiation and facilitation between plant species which can raise productivity beyond that which is achieved when species are grown alone. Differences in resource requirements by species and positive interactions between them (Mulder *et al.* 2001) is thought to allow many species to coexist and lead to greater productivity at higher diversity levels (Tilman *et al.* 1997b). Alternatively, stochastic processes involved in community assembly may have a dominant role. During experimental studies, random sampling of species within plots can represent these stochastic processes. Sampling of highly dominant species can result in increased primary production (Huston 1997). In this case, increases in productivity are not driven by increases in diversity per se, but rather by the increased likelihood of sampling highly productive species (Aarssen 1997, Wardle 1999). Despite debate regarding uncertainties in methodology it is clear that biodiversity serves an important role in buffering ecosystem productivity against harmful changes.

3 Analytical methods and data sources

3.1 Temporal analysis

A temporal analysis of pathways 1, 2, 4, 5/9, 12/14 and 15 in Fig. 1 was carried out as specified by the following system of equations[28]:

Population: $\dot{\mathbf{P}} = \boldsymbol{\rho}_P\ \mathbf{P}$ (1)

Per-capita consumption (GNE): $\dot{\mathbf{y}} = \boldsymbol{\rho}_y\ \mathbf{y}$ (2)

Per-capita gross output (GNT): $\mathbf{x} = (\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}$ (3)

Agricultural productivity (yield):
$$\vec{\nabla}\mathbf{p} = \dot{\mathbf{p}} + \vec{\nabla}_S\mathbf{p} + \vec{\nabla}_T\mathbf{p} = \boldsymbol{\rho}_L\mathbf{p} + \eta_t\frac{\mathbf{p}}{t-1990} + \eta_S\mathbf{p}\hat{\mathbf{S}}^{-1} + \Delta\mathbf{p}(T) \quad (4)$$

Land use:
$$\vec{\nabla}\mathbf{L}_j = \vec{\nabla}_P\mathbf{L}_j + \vec{\nabla}_x\mathbf{L}_j + \vec{\nabla}_o\mathbf{L}_j + \vec{\nabla}_p\mathbf{L}_j = \mathbf{L}_j\hat{\mathbf{P}}^{-1} + \mathbf{L}_j\hat{\mathbf{x}}^{-1} + (\eta_o-1)\mathbf{L}_j\hat{\mathbf{x}}^{-1} + \mathbf{L}_j\hat{\mathbf{p}}_j^{-1} \quad (5)$$

CO_2 efficiency of energy use:
$$\vec{\nabla}\mathbf{c} = \dot{\mathbf{c}} + \vec{\nabla}_x\mathbf{c} = \eta_e\frac{\mathbf{c}}{t-1990} + \vec{\nabla}_x\left(a\mathbf{x}^2 + b\mathbf{x} + c\right) \quad (6)$$

Greenhouse gas emissions:
$$\vec{\nabla}\mathbf{G}_{j=\text{fuel}} = \vec{\nabla}_P\mathbf{G}_j + \vec{\nabla}_x\mathbf{G}_j + \vec{\nabla}_e\mathbf{G}_j + \vec{\nabla}_c\mathbf{G}_j = \mathbf{G}_j\hat{\mathbf{P}}^{-1} + \eta_{g,j}\mathbf{G}_j\hat{\mathbf{x}}^{-1} + \mathbf{G}_j\dot{\mathbf{c}}_j^{-1}$$
$$\vec{\nabla}\mathbf{G}_{j=\text{livestock}} = \vec{\nabla}_P\mathbf{G}_j + \vec{\nabla}_x\mathbf{G}_j = \mathbf{G}_j\hat{\mathbf{P}}^{-1} + \eta_{g,j}\mathbf{G}_j\hat{\mathbf{x}}^{-1} \quad (7)$$
$$\mathbf{G}_{\text{forestry}} = -p_{\text{nat.forest}}L_{\text{nat.forest}} - p_{\text{plantation}}L_{\text{plantation}}$$
$$\mathbf{G}_{\text{LUC}} = -\int_{t_o}^{t} f_{\text{LUC}}\Phi_f(t-t')\dot{L}_{\text{nat.forest}}(t')dt'$$

Temperature anomaly:
$$T_j(t) = \frac{\sigma_j\beta_j}{C}\int_{-\infty}^{t}\int_{-\infty}^{t'} G_j(t'')\left(\sum_r f_{jr}e^{-\frac{t'-t''}{\tau_{jr}}}\right)\left(\sum_r\frac{l_s}{\tau_s}e^{-\frac{t-t'}{\tau_s}}\right)dt''dt' \quad (8)$$

Percentage of threatened species: $\mathbf{S} = \chi\left(\mathbf{X}\hat{\mathbf{S}}\right) + \sum_j \lambda_j\mathbf{L}_j\hat{\mathbf{L}}^{-1} + \mathbf{S}(\mathbf{T})$ (9)

Biocapacity:
$$\mathbf{B} = \frac{1}{\bar{p}}\sum_{\text{all } j}\mathbf{L}_j\hat{\mathbf{p}}_{0j}\hat{\mathbf{p}} \quad (10a)$$
$$\mathbf{B} = \sum_{\text{all } j}\mathbf{L}_j\hat{\mathbf{p}}_{0j}\hat{\mathbf{p}} \quad (10b)$$

Ecological Footprint (production):
$$\mathbf{E}^{(p)} = \frac{1}{\bar{p}}\sum_{j\text{ in use}}\mathbf{L}_j\hat{\mathbf{p}}_{0j}\hat{\mathbf{p}} \quad (11a)$$
$$\mathbf{E}^{(p)} = \sum_{j\text{ in use}}\mathbf{L}_j\hat{\mathbf{p}}_{0j}\hat{\mathbf{p}} \quad (11b)$$

Ecological Footprint (consumption): $\mathbf{E}^{(c)} = \mathbf{q}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y} = \mathbf{E}^{(p)}\hat{\mathbf{x}}^{-1}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}$ (12)

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[28] All bold variables are vectors over 239 countries (see list in Appendix), a "˙" (dot) on top of a variable (for example $\dot{Z}$) denotes a temporal derivative, and the gradient symbol $\vec{\nabla}_x$ stands for the vector $\{\partial/\partial x_i\}$. Our model is similar in its structure and formulation to previously employed models, for example the more detailed micro-model of biodiversity and land use by Eppink *et al.* 2004.

This system of equations contains:

1. Eqs. 1 and 2 are 1st-order differential equations for the standard exponential-growth model with variable growth rates, applied to population **P** and to per-capita consumption **y** in \$PPP/capita. Population forecasts ρ_P up to 2050 for evaluating Eq. 1 are published by the U.S. Census Bureau 2006. Historical GDP growth ρ_y rates are available from the United Nations Statistics Division 2007b. Both equations represent exogenous drivers in terms of the entry (pressure) point in Fig. 1.
2. Eq. 3 is the standard Leontief quantity model (Leontief 1986), deducing gross output responses as a result of final consumption changes. **x** holds gross output (Gross National Turnover) in \$PPP/capita; **A** is the direct requirements matrix of a multi-region input-output table of the world economy, and **I** is the identity matrix. Note that all assumptions of classical input-output theory apply, such as absence of capacity constraints, homogeneity of production sectors, fixed production recipe, etc. **x**, **A** and **y** were constructed from data on Gross Domestic Product by country (Central Intelligence Agency 2006) and on international commodity trade (United Nations Statistics Division 2007a).
3. Agricultural productivity (Eq. 4) was modeled a unit-less index to 2005 states, using a) a linear time trend with country-specific rates ρ_L describing declining productivity as a result of ongoing land use (path 15), b) a non-linear saturating time trend with a uniform elasticity η_t describing technological improvements over time (exogenous, see Section 2.1.3), c) a η_S–elastic response to biodiversity changes (path 12/14, see Section 2.7), and d) a linear estimate for the effect of climate change of productivity (path 13). Yield declines ρ_L and initial states as of 2005 were constructed from country estimates of productivity reductions (land and soil degradation) due to deforestation, overexploitation, erosion and pollution (GLASOD weights in International Soil Reference and Information Centre (ISRIC) and United Nations Environment Programme 2000). η_t is estimated in Fig. 6. $\eta_S = 0.173$ was taken from Costanza *et al.* 2007, who take non-threatened species density as a proxy for the biodiversity variable in Tilman 1999. The linear relationship between temperature anomaly T and productivity **p** was modeled using a $\mathrm{d}p/p$ – $\mathrm{d}T/T$ coefficient that decreases from 6%/ºC at 0.5ºC temperature anomaly to 0 at 3ºC temperature anomaly, which is a compromise between the studies of Morales *et al.* 2007 (3%/ºC) and Cramer *et al.* 2001 (10%/ºC).
4. Eq. 5 describes requirements of productive land **L** of type j for populations **P** producing gross output **x**, in units of hectares, using an elasticity η_o in order to link percentage changes in per-capita gross output to percentage changes of land use (path 1). $\mathbf{L} = \sum_j \mathbf{L}_j$ is total land use. Increases of productive land come at the cost of unoccupied non-permanent arable land (assumed 50% of total non-permanent land at 2005), and natural forest. Data on 2005 land use $\mathbf{L}_j$ of type j by country was sourced from on-line databases of the FAO Statistics Division (FAOSTAT) 2004, and from the Global Forest Assessment database (FAO Statistics Division (FAOSTAT) 2001), the latter relying on satellite imagery (Global Land Cover Facility (GLCF) 2002).[29] η_o was taken from Figs. 4 and 5. For countries where recent land clearing rates were available (from Task Force on National Greenhouse Gas Inventories 1996, Tab. 5-4), η_o was adjusted so our model would reproduce these rates. This adjustment leads to a downward adjustment of η_o for grazing land, and an upward adjustment of η_o for permanent crops, thus naturally reflecting the response of countries facing land

[29] Non-arable land was calculated as the remainder of total land area and land types.

limitations in switching from pasture livestock towards feedlot and monogastrics plus associated feed crop production. Note that we have ignored trade responses to land limitations, with the result that some countries, especially small island states, simply "run out" of land before 2050.

5. Eq. 6 describes the combined effect **c** of technological improvements in energy efficiency, and changes in the carbon content of fuels through fuel mix changes, indexed to 2005. For energy intensities, we use country-specific η_e–elastic time trends (modeled from data in World Resources Institute 2006, see Figs. 9-11), and for the carbon content we assume a uniform Kuznets relationship with coefficients a, b and c (Fig. 12).
6. Eq. 7 describes emissions $\mathbf{G}_j$ of greenhouse gas j for populations **P** producing gross output **x**, in units of tonnes (paths 2 and 4). For CO_2 emissions from fuel combustion and CH_4 and N_2O emissions from livestock (enteric fermentation and manure), we use an elasticity η_g in order to link percentage changes in per-capita gross output to percentage changes in emissions (path 2). CO_2 sequestration of commercial and non-commercial forests is coupled to the land area changes of plantations and natural forest. Within land use changes, we consider CO_2 emissions during land clearing (burning of above-ground biomass, decay of above-ground biomass over 10 years, and below-ground exponential releases of soil carbon over 20 years), and model decays and uptakes with convolution integrals containing normalised response functions Φ for the various processes. Initial 2000 emissions were constructed from fuel combustion CO_2 data (International Energy Agency 2004), from country-level livestock unit numbers by animal type (Livestock Information and Policy Branch 2007), and from region-specific emission factors for enteric fermentation and manure, and for forest burning, decay and soil carbon (Task Force on National Greenhouse Gas Inventories 1996). For CH_4 and N_2O emissions we assumed $\eta_g = \eta_o$, while for fuel combustion Fig. 8 shows $\eta_g(x) = 0.963 - 0.022 \ln(x)$. Land use change factors f_{LUC} for forest burning, decay and soil carbon were taken from IPCC Guidelines (Task Force on National Greenhouse Gas Inventories 1996). Land use changes and forestry source and sink factors were weighted by country with ecosystem type percentages (World Resources Institute 2005b) in order to account for the type of natural vegetation burned or sequestering.
7. Eq. 8 was taken straight out of the Brazilian proposal to the IPCC (Meira and Miguez 2000, Eq. 14). It links the projected global temperature anomaly T_j (in ºC) to emissions G_j of greenhouse gas j. C is the heat capacity of the Earth's climate system, σ_j is the change in radiative forcing per unit of additional concentration of greenhouse gas j, β_j is the increase in the concentration of greenhouse gas j per unit of annual emissions of that gas, r counts the number of atmospheric decay fractions (weighted f_{jr}) of the same gas j with different lifetimes τ_{jr}, and s counts the number of climate response fractions (weighted l_s) with different lifetimes τ_s. Eq. 8 was evaluated using emission profiles resulting from Eq. 7, as well as constants taken from Rosa *et al.* 2004 (CO_2), Blasing and Smith 2006 (CH_4 and N_2O), and Houghton *et al.* 1996 (all gases). The average global temperature change resulting from summing T_j over all greenhouse gases j was finally distributed across countries (yielding a vector $\Delta\mathbf{T}$) based on climate model results from Cubasch *et al.* 2001.

8. Eq. 9 describes the percentage of threatened species[30] as a function of land use pattern and threatened species in neighbouring countries (path 5/9), as well as temperature change due to global warming (path 6/9). The first component takes the form of a spatial lag model, as used for example by McPherson and Nieswiadomy 2005; Pandit and Laband 2007a; b. χ and λ_i are characterisation factors for spatial autocorrelation and land use. **X** is a normalised adjacency matrix containing percentages of shared borders (Cliff and Ord 1981), and $\mathbf{L}_j\hat{\mathbf{L}}^{-1}$ is the percentage land use pattern. Data on the number of threatened plants, mammals, birds, reptiles, amphibians and fish (**S**) were taken from The IUCN Red List of Threatened Species 2006. Threatened species are those listed as “Critically Endangered” (CR), “Endangered” (EN) or “Vulnerable” (VU).[31] All species numbers were converted into percentages, using the number of total species recorded in each of the taxonomic groups, as reported in the World Resources Institute’s Biodiversity Overview (World Resources Institute 2005a)[32]. The autoregression coefficient χ was taken from multiple regression analyses in Lenzen *et al.* 2007b, and characterisation factors λ_i are an average over LCA studies as documented in Tab. 2. The adjacency matrix **X** was constructed by simply placing the border length in kilometres of each country *i* with all its neighbours *j* in cells X_{ij}.[33] Information on the length of international borders and coast lines was taken from the CIA’s World Factbook (Central Intelligence Agency 2006). The second component was estimated by regressing data on relationship between global mean annual warming and the fractional global area with novel and disappearing climates (Williams *et al.* 2007). For simplicity we assume a linear relationship between the proportion of land area with disappearing climates and the proportion of species at risk. A power relationship ($S = Area^z$) might be more appropriate. Although *z* values can be calculated from country-level information on species numbers and land area, the shape of the species area curve for different taxa at the global scale is uncertain. Note that we have added changes in both components, because of a lack of information on the overlap of land us and climate change effects.

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[30] Strictly speaking, the species portion we relate to disappearing climates does not adhere to the IUCN definition of a "threatened" species. Our combined measure **S** is perhaps better termed as "species at risk". However, for the sake of simplicity, we retain the IUCN term.

[31] The number of threatened species excludes introduced species, species whose status is insufficiently known (categorized by IUCN as “data deficient”), those known to be extinct, and those for which status has not been assessed (categorized by IUCN as “not evaluated”). Species are classified as vulnerable or endangered if they face a risk of extinction in the wild in the immediate future (critically endangered), in the near-term (endangered), or in the medium-term (vulnerable). Threat categories are assigned based on total population size, distribution, and rates of decline (The IUCN Red List of Threatened Species 2006). Note that cetaceans are not assigned to particular countries except for inshore or coastal species, and are therefore missing from this analysis. Similarly, only vascular plants are considered, only land-nesting or –breeding marine species are counted, and marine turtles and most marine fish are excluded (cf. Naidoo and Adamowicz 2001).

[32] The Total Number of Known Species refers to the total number of a particular type of species in a given country. Data on known mammals exclude marine mammals. Data on known birds include only birds that breed in that country, not those that migrate or winter there. The number of known plants includes higher plants only: ferns and fern allies, conifers and cycads, and flowering plants. The number of known species is collected by the United Nations (UN) Environment Programme-World Conservation Monitoring Centre (UNEP-WCMC) from a variety of sources, including, but not limited to, national reports from the Convention on Biodiversity, other national documents, independent studies, and other texts. Data are updated on a continual basis as they become available; however, updates vary widely by country. While some countries (WCMC estimates about 12) have data that were updated in the last six months, other species estimates have not changed since the data were first collected in 1992.

[33] A simpler alternative is a binary contiguity matrix, which just contains elements valued 1 for pairs of bordering countries, and 0 otherwise.

9. Eq. 10 describes biocapacity as available annual biomass production for human purposes, in units of global hectares (10a) or tonnes (10b), from all land types. $\hat{\mathbf{p}}_{0j}$ holds 2005 yields (in units of t/ha, not indexed) as measured for land type j, derived from data in FAO Statistics Division (FAOSTAT) 2007b, Livestock Information and Policy Branch 2007, FAO Statistics Division (FAOSTAT) 2007c, and FAO 2007. We express biocapacity in global hectares as well as in physical mass units. Conversion between the two proceeds simply by dividing by world-average yield $\bar{p}$ (average over all countries and all land types). The vector $\mathbf{p}_0/\bar{p}$ holds the product of country- and crop-specific yield and equivalence factors (see Wackernagel *et al.* 2005).
10. The Ecological Footprint of production in Eq. 11 is a subset of biocapacity in Eq. 10, spanning only areas used by humans, thus excluding natural forest and unoccupied non-permanent arable land from the sum. Once again, it can be expressed either in global hectares (11a) or tonnes (11b). Note that fisheries are excluded from our analysis. While in the static Ecological Footprint method fish protein is added as an equivalent of terrestrial livestock-based protein, it cannot be included in the same way in this dynamic approach, because the magnitude of fish harvests does not impinge on land requirements.
11. The Ecological Footprint of consumption in Eq. 12 is calculated using once again the standard Leontief quantity model (Leontief 1986), deducing Ecological Footprint requirements of final consumption levels $\mathbf{y}$. The terms $\mathbf{q} = \mathbf{E}^{(p)}\hat{\mathbf{x}}^{-1}$ represent direct Ecological Footprint intensities, while the $\mathbf{E}^{(p)}\hat{\mathbf{x}}^{-1}(\mathbf{I} - \mathbf{A})^{-1}$ are called multi-regional Leontief multipliers.

Our analysis was carried out on a set of 239 countries, which are listed in the Appendix. Differential equations were solved iteratively by using a simple Laspeyres-type finite-difference scheme, for example for Eq. 5:

$$\begin{aligned} L_{ij,t} &= L_{ij,t-1} + \frac{\partial L_{ij,t}}{\partial P_{i,t}} \Delta P_{i,t} + \frac{\partial L_{ij,t}}{\partial p_{i,t}} \Delta p_{i,t} \frac{\partial L_{ij,t}}{\partial o_{i,t}} \Delta o_{i,t} + \frac{\partial L_{ij,t}}{\partial p_{i,t}} \Delta p_{i,t} \\ &= L_{ij,t-1} + \frac{L_{ij,t-1}}{P_{i,t-1}} \left[P_{i,t} - P_{i,t-1}\right] + \eta_x \frac{L_{ij,t-1}}{x_{i,t-1}} \left[x_{i,t} - x_{i,t-1}\right] + \frac{L_{ij,t-1}}{p_{i,t-1}} \left[p_{i,t} - p_{i,t-1}\right] \end{aligned}, \tag{13}$$

or for Eq. 9:

$$S_{i,t} = \chi\left(\sum_{k=1}^{N} X_{ik} S_{k,t-1}\right) + \sum_j \lambda_j L_{j,t-2} L_{t-2}^{-1} . \tag{14}$$

3.2 Monte-Carlo simulation

The sensitivity of the system of Eqs. 1-9 with respect to parameter uncertainty was tested by carrying out a Monte-Carlo simulation.[34] We assume that stochastic uncertainty ranges can be specified for ρ_P, ρ_y, ρ_L, η_t, η_S, η_o, η_e, η_c, $\eta_{g,j}$, χ, f_{LUC}, and $\boldsymbol{\lambda}$. Standard deviations $\boldsymbol{\sigma}_\rho$, $\boldsymbol{\sigma}_\eta$, etc were estimated from ranges documented in the literature. 250 simulations were carried out,

[34] See Bullard and Sebald 1977; 1988; Lenzen 2001 for further details.

where all parameters were perturbed stochastically, and the system of Eqs. 1-12 was solved again. This technique yields a bundle of possible scenarios, which are concentrated around a most likely trajectory.

3.3 Structural Decomposition Analysis

Structural Decomposition Analysis (SDA) aims at separating a time trend of an aggregate measure into a number of influencing driving forces, which can be accelerators or retardants (Dietzenbacher and Los 1998; Hoekstra and van den Bergh 2002). The basic approach to additive structural decompositions of a function $y(x_1, x_2, ..., x_n)$ of n determinants is through its total differential

$$\mathrm{d}y = \frac{\partial y}{\partial x_1}\mathrm{d}x_1 + \frac{\partial y}{\partial x_2}\mathrm{d}x_2 + ... + \frac{\partial y}{\partial x_n}\mathrm{d}x_n \ . \tag{15}$$

In the most common case, $y(x_1, x_2, ..., x_n) = x_1 \cdot x_2 \cdot ... \cdot x_n$ (with the x_i being scalars, vectors or matrices), so that

$$\mathrm{d}y = \prod_{j=1, j\neq 1}^{n} x_j \mathrm{d}x_1 + \prod_{j=1, j\neq 2}^{n} x_j \mathrm{d}x_2 + ... + \prod_{j=1, j\neq n}^{n} x_j \mathrm{d}x_n = \sum_{i=1}^{n}\left(\prod_{j=1, j\neq i}^{n} x_j \,\mathrm{d}x_i\right) . \tag{16}$$

In our case, y represents the Ecological Footprint as in Eq. 9, and its determinants x_i are – amongst others – yields, biodiversity, per-capita consumption, and population. In practice, there are a number of variants that solve Eqs. 15 and 16, such as Laspeyres- and Divisia-type decompositions (Ang 2004). In this work, we apply a Paasche-type additive decomposition of the form

$$\Delta y^{\mathrm{P}} = \sum_{i=1}^{n}\left(\prod_{j=1, j\neq i}^{n} x_j^{(t+1)}\left(x_i^{(t+1)} - x_i^{(t)}\right)\right). \tag{17}$$

Eq. 17 demonstrates the principle of a typical structural decomposition: The *ceteris-paribus terms* $\prod_{j=1, j\neq i}^{n} x_j^{(t+1)}\left(x_i^{(t+1)} - x_i^{(t)}\right)$ measure the effect on y if only the determinant x_i changed at any one time.[35] Like other Laspeyres-type decompositions, Paasche decompositions are never exact, that is the effect on y of *simultaneous* changes in the x_i is contained in *residuals* $\Delta y - \Delta y^{\mathrm{P}} \neq 0$, also referred to as "joint" or "interaction" terms (Lenzen 2006a, De Bruyn 2000 Sec. 9.4, and Casler 2001 p.146–148).[36]

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[35] For example, a ceteris-paribus term for $y = x_1 x_2$ of determinant x_1 is $x_2^{(1)} \Delta x_1$, where x_2 stays constant during the change in x_1.

[36] There exist exact decomposition methods (Dietzenbacher and Los 1998; Sun and Ang 2000; Albrecht *et al.* 2002). However, the sizeable differences between residuals and ceteris-paribus terms that can occur between different non-exact decomposition methods, suggests a kind of arbitrariness in trying to allocate these residuals to any one of the determinants in order to achieve an exact decomposition. One might even argue that there is little trade-off between this arbitrariness and the "unexplained-ness" of residuals, and therefore not much gained in preferring an exact over a non-exact structural decomposition method.

3.4 Multi-region trade balances

In a series of static analyses, threatened species S_i and Ecological Footprints E_i were allocated to the production (*GDP*) of each country, as well as consumption (Gross National Expenditure, *GNE*), exports (*X*), and imports (*M*). Trade balances $\Delta T = X - M$ were evaluated using the National Accounting identity $GDP + M = GNE + X$. Values for ΔT, X, and M by country were constructed from data on Gross Domestic Product by country (Central Intelligence Agency 2006) and on international commodity trade (United Nations Statistics Division 2007a).

3.5 Structural Path Analysis

Structural Path Analysis (SPA[37]) were performed on the generalised multi-region input-output system, in order to identify the most important international trade flows in terms of threatened species and the gap between the Ecological Footprint and biocapacity.

SPA employs the direct requirements matrix **A** in order to decompose multi-regional multipliers $\mathbf{E}^{(p)}\hat{\mathbf{x}}^{-1}(\mathbf{I}-\mathbf{A})^{-1}$ into contributions from single international supply chains (so-called "structural paths"), by "unraveling" Eq. 12 using the series expansion of the Leontief inverse:

$$\mathbf{q}\,(\mathbf{I}\text{-}\mathbf{A})^{-1}\,\mathbf{y} = \mathbf{qy} + \mathbf{qAy} + \mathbf{qA}^2\mathbf{y} + \mathbf{qA}^3\,\mathbf{y} + \ldots \quad . \tag{18}$$

Expanding Eq. 18, the Ecological Footprint can be written as (Lenzen 2002; 2006b)

$$\begin{aligned} E^{(c)} &= y_i \sum_{j=1}^{n} q_j \left(\delta_{ji} + A_{ji} + (\mathbf{A}^2)_{ji} + (\mathbf{A}^3)_{ji} + \ldots\right) \\ &= y_i \sum_{j=1}^{n} q_j \left(\delta_{ji} + A_{ji} + \sum_{k=1}^{n} A_{jk}A_{ki} + \sum_{l=1}^{n}\sum_{k=1}^{n} A_{jl}A_{lk}A_{ki} + \ldots\right) \\ &= q_i y_i + \sum_{j=1}^{n} q_j A_{ji} y_i + \sum_{k=1}^{n} q_k \sum_{j=1}^{n} A_{kj}A_{ji} y_i + \sum_{l=1}^{n} q_l \sum_{k=1}^{n} A_{lk} \sum_{j=1}^{n} A_{kj}A_{ji} y_i + \ldots \quad , \end{aligned} \tag{19}$$

where i, j, k, and l denote industries, and δ_{ij}=1 if i=j and δ_{ij}=0 otherwise. The total Ecological Footprint is thus a sum over a direct footprint $q_i y_i$, occurring in country i itself, and higher-order input paths. An input path from country j into country i of first order is represented by a product $q_j A_{ji} y_i$, while an input path from country k via country j into country i is represented by a product $q_k A_{kj} A_{ji} y_i$, and so on. For an n-country table **A**, there are n input paths of first order, n^2 paths of second order, and, in general, n^N paths of N^{th} order.

[37] SPA was introduced into economics and regional science in 1984 as a general decomposition approach (Crama *et al.* 1984; Defourny and Thorbecke 1984), and later applied in life-cycle assessment by (Treloar 1997; Lenzen 2002; Lenzen and Treloar 2002).

4 Forecasting the Ecological Footprint of Nations

The analytical outcomes will be presented in a causal chain that looks first at the acceptability of the emissions and global change part of the framework and then at land use change, the footprint and biocapacity changes due to population growth and economic growth, the effect of these driving forces on species loss by country, the effect of world trade and finally at the biocapacity-Ecological-Footprint gap, and the rate of its closure. In these result areas, the structural relationships of the complex inter-related data set will be explored using structural path and structural decomposition analyses. Due to uncertainties in current and future data, Monte-Carlo analysis is performed to provide an idea of the uncertainty of the global outcomes.

4.1 Climate analysis

Before proceeding with land use and Ecological Footprint analyses, we checked the results of our climate projections against historical measurements published in the literature (Fig. 13). In order to realistically forecast future greenhouse gas concentrations, a 300-year history of emissions (back to pre-industrial times) is required because of the long residence time of CO_2. Similarly, in order to forecast temperature, a 300-year history of greenhouse gas concentrations is required. Our simplified model of CO_2 emissions reflects well historical emissions (Houghton 2003; Marland *et al.* 2006), especially for the more recent period between 1970 and 2005 (left graph in Fig. 13). However, our bottom-up analysis of CH_4 emissions from livestock reproduces only 80% of emissions reported by Stern and Kaufmann 1998. More recent estimates (Lassey 2007**)** appear to confirm our lower figure of around 80 Mt CH_4.

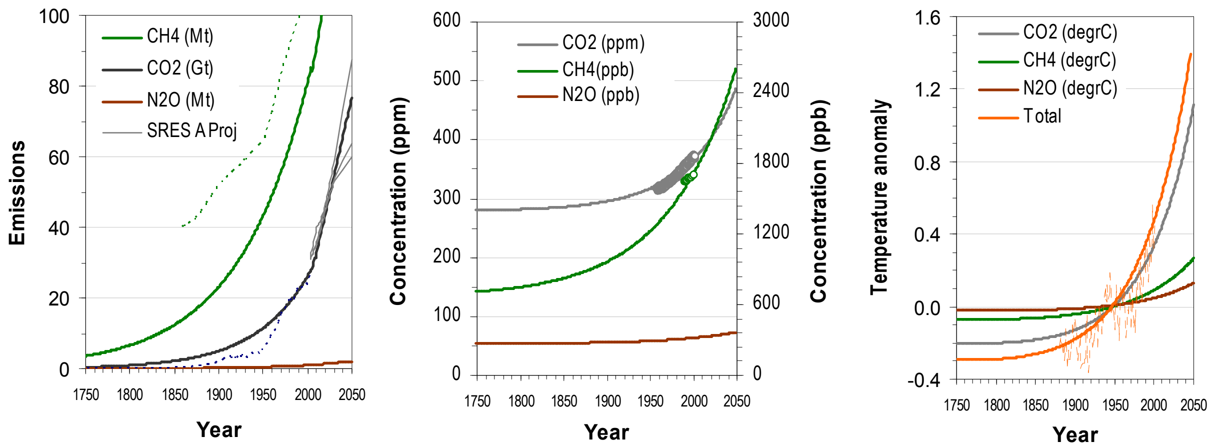

Fig. 13: Greenhouse gas emissions, concentrations, and temperature anomalies resulting from evaluating the Brazilian proposal in Eq. 8 (thick solid lines), historical measurements (dotted lines and circles), and projections (thin solid lines).

Evaluating Eq. 8 with these emissions profiles (middle graph) reproduces the magnitude of available – albeit short – concentration measurements, from the Mauna Loa Observatory (dotted grey line, CO_2, Keeling and Whorf 2005), and from the CSIRO Macquarie Island

Sampling Station (dotted green line, CH_4, Steele *et al.* 2003). Similarly, temperature anomalies computed by NASA (Hansen *et al.* 2006) are reasonably well reproduced (dotted orange line, right graph). Given the uncertainties of remaining elements in our analysis, we feel that the accuracy of the modeled climate parameters is sufficient.

Accepting the broad reproducibility of historical climate trends as documented in Fig. 13, the model predicts global greenhouse gas emissions from fuel combustion, land use change, forestry, and livestock to more than double from about 30 Gt CO_2-e in 2005 to about 80 Gt CO_2-e in 2050. This increase is mainly due to increases in CO_2 resulting from the combustion of fossil fuels, with the main contributions from North America and Asia (Fig. 14). Major contributions from land use changes in Latin America, Africa and Asia are projected to peak around 2030 (see next Section). Our projections are in reasonable agreement with SRES marker scenarios of the type A (Fig. 13, left graph, thin solid grey lines, after Intergovernmental Panel on Climate Change 2000a).[38]

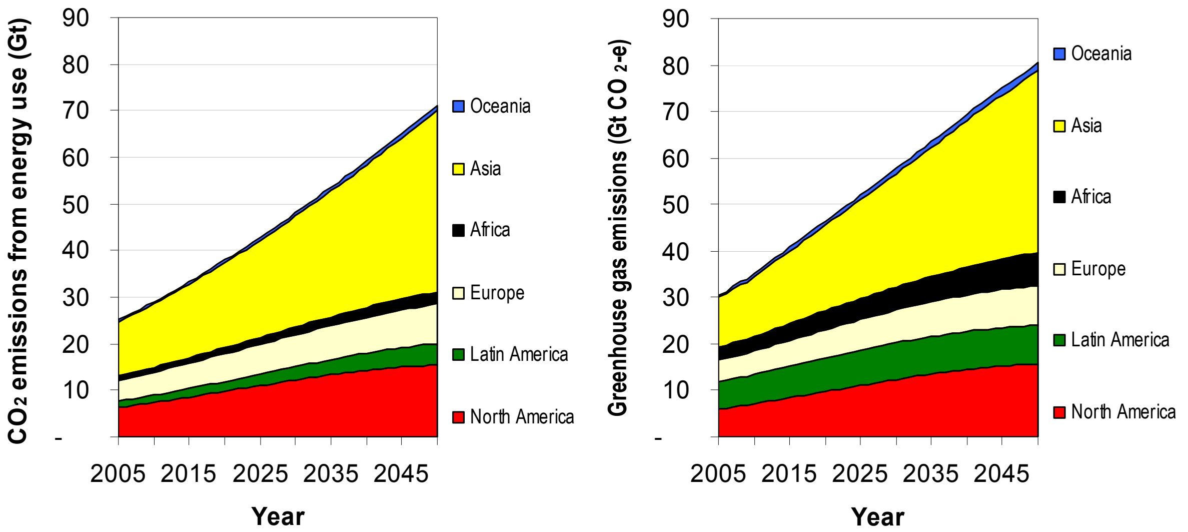

Fig. 14: Projected CO_2 emissions from fossil fuel combustion, and CO_2-equivalent emissions from fuel combustion, land use change, forestry, and livestock.

[38] According to the Intergovernmental Panel on Climate Change 2000b, "the A1 storyline and scenario family describes a future world of very rapid economic growth, global population that peaks in mid-century and declines thereafter, and the rapid introduction of new and more efficient technologies. Major underlying themes are convergence among regions, capacity building and increased cultural and social interactions, with a substantial reduction in regional differences in per capita income. The A1 scenario family develops into three groups that describe alternative directions of technological change in the energy system. The three A1 groups are distinguished by their technological emphasis: fossil intensive (A1FI), non-fossil energy sources (A1T), or a balance across all sources (A1B) (where balanced is defined as not relying too heavily on one particular energy source, on the assumption that similar improvement rates apply to all energy supply and end use technologies). […] The A2 storyline and scenario family describes a very heterogeneous world. The underlying theme is self-reliance and preservation of local identities. Fertility patterns across regions converge very slowly, which results in continuously increasing population. Economic development is primarily regionally oriented and per capita economic growth and technological change are more fragmented and slower than in other storylines."

4.2 Analysis of Ecological Footprints and their drivers

Results for the globe are presented in four graphs (see Fig. 15 for the world level): The top two graphs represent drivers, and the bottom two various representations of biocapacity and the Ecological Footprint. Clockwise from top left, they are: 1) land use pattern (in % of total, top left), 2) selected variables from Eqs. 1-9 (population, per-capita GNE, productivity, % threatened species, and greenhouse gas emissions, top right), 3) total Ecological Footprint and biocapacity (in units of "2005 Earth's"), and gap between them (in millions of 2005gha, bottom left), and 4) per-capita biocapacity and Ecological Footprint (in units of gha, 2005gha, and tonnes, bottom right).

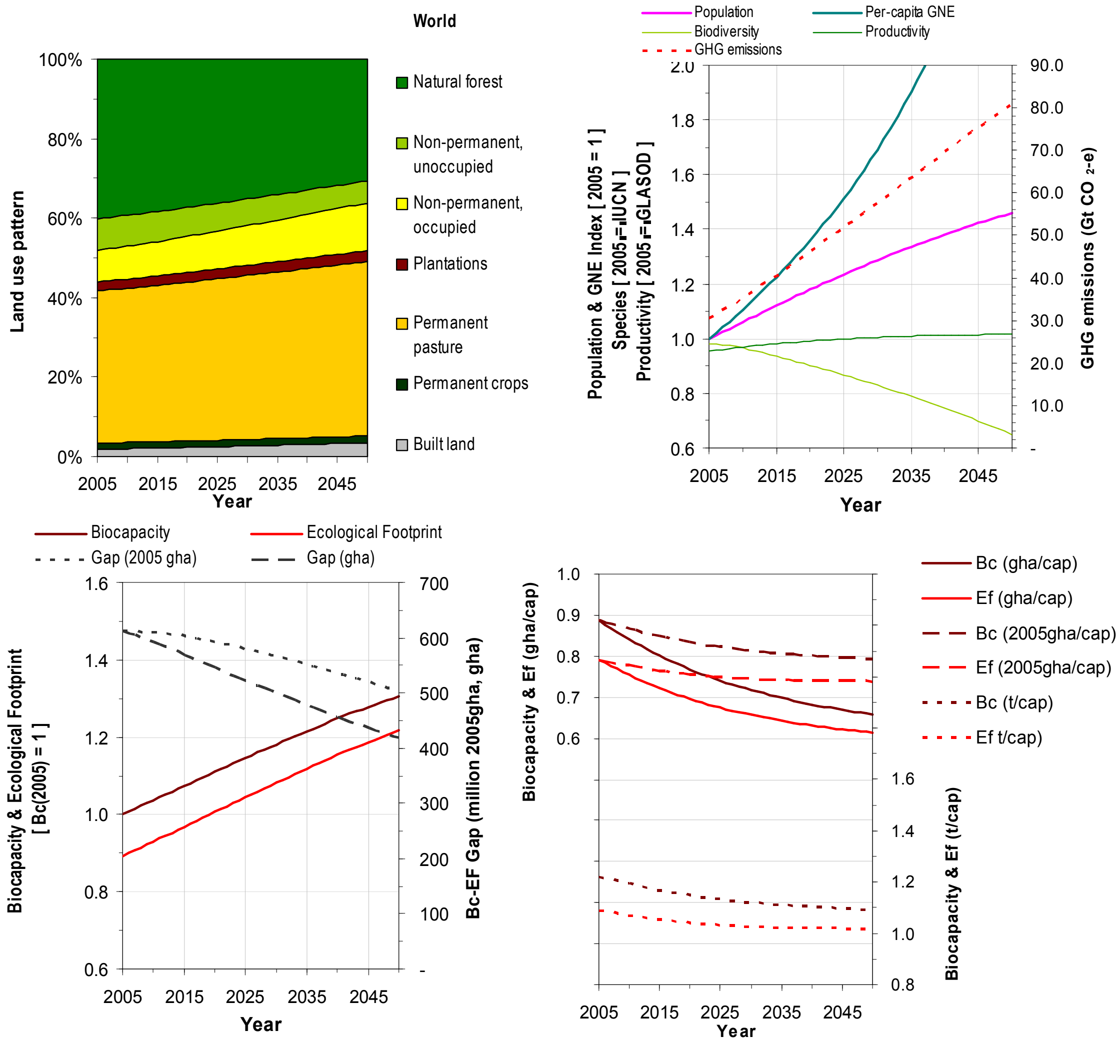

Fig. 15: Aggregation of temporal analyses of all countries.

Based on the parametrisation of Eqs. 1-9, the agricultural and built landscape is projected to expand from roughly 50% of total terrestrial area to about 60% in 2050, at the cost of unused non-permanent arable land (assumed to be 50% of all non-permanent arable land in 2005), and natural forest (Fig. 15, top left).[39] This land use expansion is underpinned by a world population growing by 50%, and by more than doubling personal wealth, thus continuing the trends in Figs. 2 and 3. Our forecasts for crop- and pastureland are only slightly (11% and 4%, respectively) below forecasts by Tilman *et al.* 2001a.[40]

In comparison, productivity increases only by about 15%, which is the combined result of improved technology (Fig. 6), land degradation (International Soil Reference and Information Centre (ISRIC) and United Nations Environment Programme 2000), biodiversity decline, and a 1.5ºC warming effect. We estimate the latter component to cause an increase in productivity of no more than 10%, which is at the lower end of estimates by Rounsevell *et al.* 2005 (+12 ±3%). Note the disproportionally high increase in non-permanent land, which is a result of a shift from grazing to feedlot production. Without this shift, the area under human appropriation would have grown more in line with population and per-capita affluence. World-wide, more than 30% of species are projected to be threatened by 2050, which is a 10-fold increase assuming no underreporting in 2005 (top right).

In unison, all trends lead to a significant (≈30%) increase in the Ecological Footprint (through increased appropriation of bioproductivity) as well as in biocapacity (through conversion of low-yield forest and unused non-permanent arable land, bottom left). The graph shows total biocapacity and the Ecological Footprint indexed to 2005 biocapacity, or in "2005 Earths".[41] For 2005, we measure the world's Ecological Footprint as about 5.1 billion global hectares (or about 7 Gigatonnes), which amounts to 0.8 global hectares per capita (or about 1 t/cap), compared to its biocapacity as 5.7 billion gha (7.8 Gt), or 0.9 gha/cap (1.2 t/cap). The difference between the two (the "gap") represents unused available bioproductivity. In 2005 it measures just over 600 million gha (about 850 Mt) and hence constitutes about 10% of total biocapacity. During the period 2005-2050 it is projected to decrease by about 90 million gha (or 150 Mt), or about 15-20% of 2005 gap width. Note that both biocapacity and the Ecological Footprint can be measured either in global hectares (Eqs. 10a and 11a) or as tonnes of vegetables, grains, milk, meat, timber, etc (Eqs. 10b and 11b).[42] The conversion between tonnes and global hectares proceed simply through dividing by world-average yield.

Displayed per-capita, biocapacity and the Ecological Footprint decrease, no matter whether they are expressed in tonnes, global hectares, or 2005 constant global hectares. The decrease in tonnes per capita is equivalent to the decrease in constant global hectares, and basically portrays a growing world population that has to make do with an ever-shrinking allotment of space and productive output for each individual. When shown in actual global hectares per capita, this decrease is accentuated by a 15% yield increase over the analysis period.

In Fig. 15, unsustainable societies demand more than can be regenerated, causing narrowing biocapacity-Ecological-Footprint gaps, that is, shrinking unused bioproductivity reserves. A

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[39] Note that we ignore structural effects in domestic production recipes and diets, world trade, as well as geographic and legal restrictions to land occupation.

[40] Tilman *et al.* 2001a (p. 282) writes that "because of the exponential nature of past global population and economic growth, we had anticipated exponential temporal trends for these [land] variables. Surprisingly, each was a linear […] function of time […]." This work confirms the quasi linear trend of agricultural land expansion, and explains this with the sub-linear relationship of agricultural output with economic growth (Section 2.1.2).

[41] Indexing to one particular year depicts Ecological Footprints in constant terms, just as GDP or other economic measures can be expressed in constant prices (see Global Footprint Network 2006).

[42] Note that we measure pasture yields as livestock (live weight) yields, and not as forage yields, because the former are documented by more extensive data than the latter. For biocapacity-Ecological-Footprint comparisons, this circumstance does not matter.

sustainable situation would have societies not demanding more than can be regenerated, so that the biocapacity and Ecological Footprint curves in Fig. 15 would run at least parallel. The gap width between biocapacity and the Ecological Footprint can be interpreted as a measure of risk under potential future production declines and/or demand increases. Whether expressed in absolute or per-capita terms, the biocapacity-Ecological-Footprint gap is experiencing a continuous closure (see Section 4.3).

Note that in a dynamic view, the appropriation of bioproductivity due to greenhouse gas emissions has to occur at the time of climate change, and not at the time of emission, because at the time of emission, bioproductivity is yet unaffected. This is a key difference to the static approach (Wackernagel *et al.* 2005) which adds the area needed to sequester global emissions on top of the actually appropriated land area, and thus creates overshoot, no matter that emissions are actually not sequestered (Wackernagel *et al.* 2002). This overshoot does hence not refer to the current world (as this world is functioning), but refers to a condition for emissions compensation that is not met in reality, or in other words a "borrowing on natural capital" (Wackernagel *et al.* 1999). Such an overshoot variable is not in any causal relationship with driver variables such as land use. In a temporally explicit, dynamic method there cannot be at any time more land used than available, but instead overshoot manifests itself in converging biocapacity-Ecological-Footprint curves, which is a vivid indicator for the fact that today's greenhouse gas emissions will have a delayed effect on future bioproductivity. In the above analogy, "the debt appears over time", and a dynamic method shows up likely trajectories.

A consequence of assessing greenhouse gas emissions at the time of their impact is that Fig. 15 refers to actual land only. However our results are in good agreement with the results published by the Global Footprint Network (GFN)[43]. Adding grazing land, crop land and harvested forest[44], the GFN shows a per-capita Ecological Footprint curve that decreases as in Fig. 15 (bottom right), and states a 2003 global Ecological Footprint of about 1 gha/cap.

4.2.1 Regional analyses

Future projections as in Fig. 15 vary significantly between world regions (Figs. 16 and 17): In North America and Europe, agricultural land is not anymore replacing natural ecosystems at significant scales (because of increasing yields and shifting to feedlots), per-capita affluence is increasing at moderate rates, and population curves are "flattening out". Biodiversity is still declining because of global climate change, however these regions are able to maintain the gap between biocapacity and the Ecological Footprint (Fig. 16). Oceania occupies a somewhat intermediate position between North America and Europe on one hand, and the suite of continents in Fig. 17 on the other. Latin America, Africa and to a lesser extent Asia are likely to feature high degrees of land conversion and associated biodiversity decline necessary for meeting the demand for bioproductivity. Thus in a territorial sense, the biocapacity stocks of North America, Europe may improve because the territorial pressure is replaced by imports, which are sourced generally from developing countries with remaining natural vegetation.

[43] http://www.footprintnetwork.org/gfn_sub.php?content=global_footprint (Fig. 3).

[44] We exclude fishing grounds since we have not assessed the indirect effects between fisheries and terrestrial land requirements. We count built land always as non-productive, but we subtract land transformed from forest or agricultural land into built land from biocapacity.

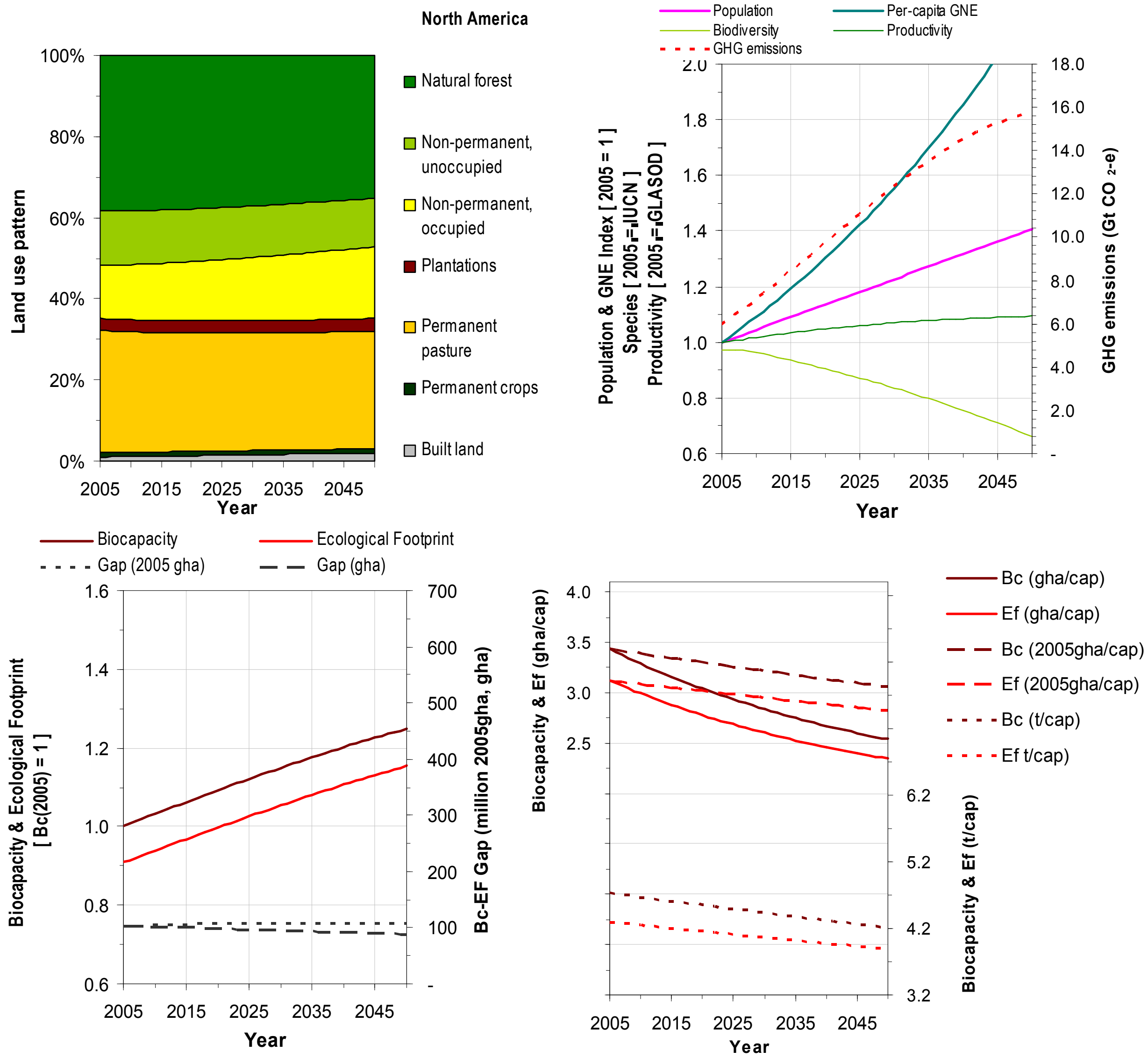

Fig. 16a: Future change in North America.

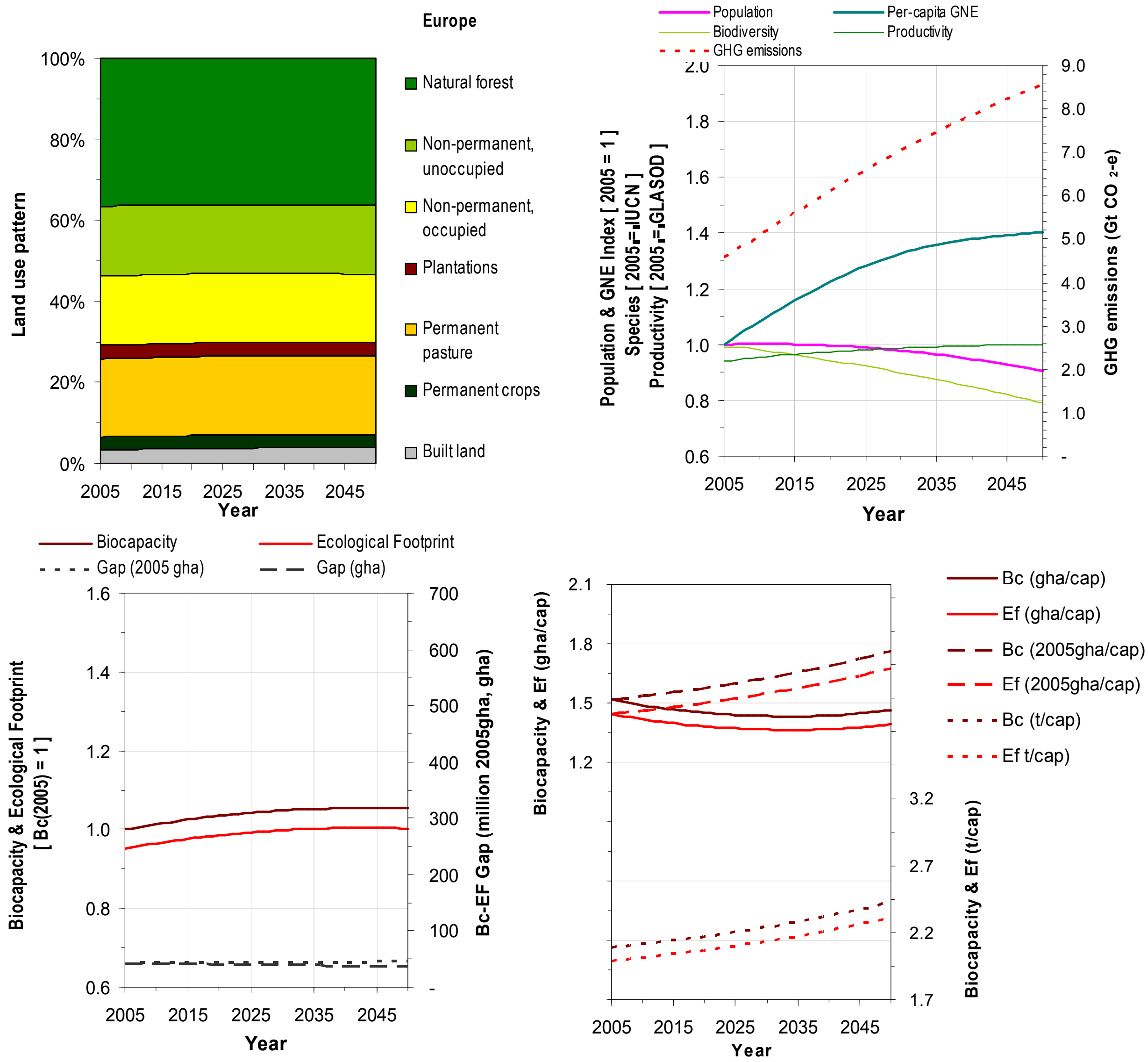

Fig. 16b: Future change in Europe.

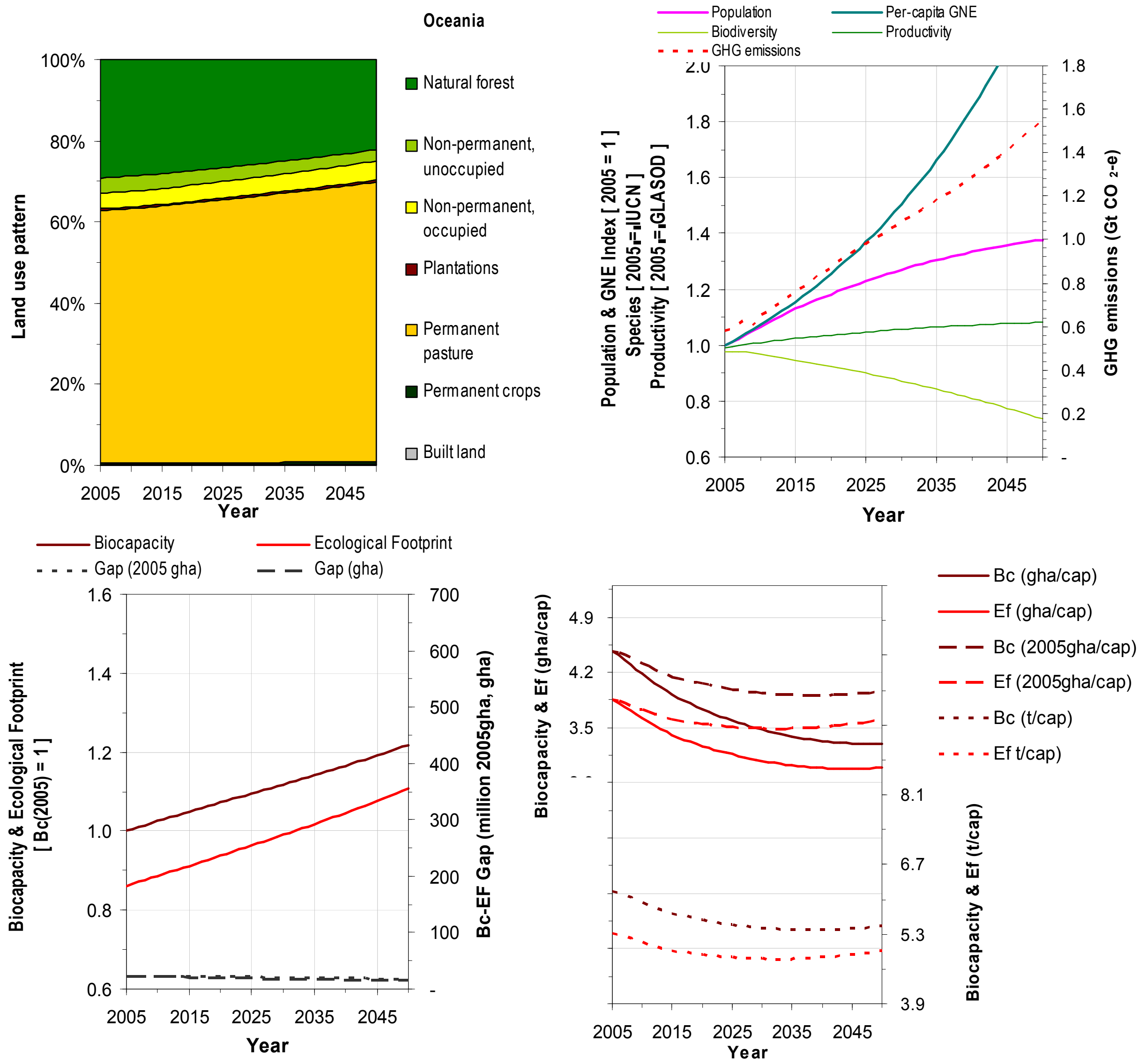

Fig. 16c: Future change in Oceania.

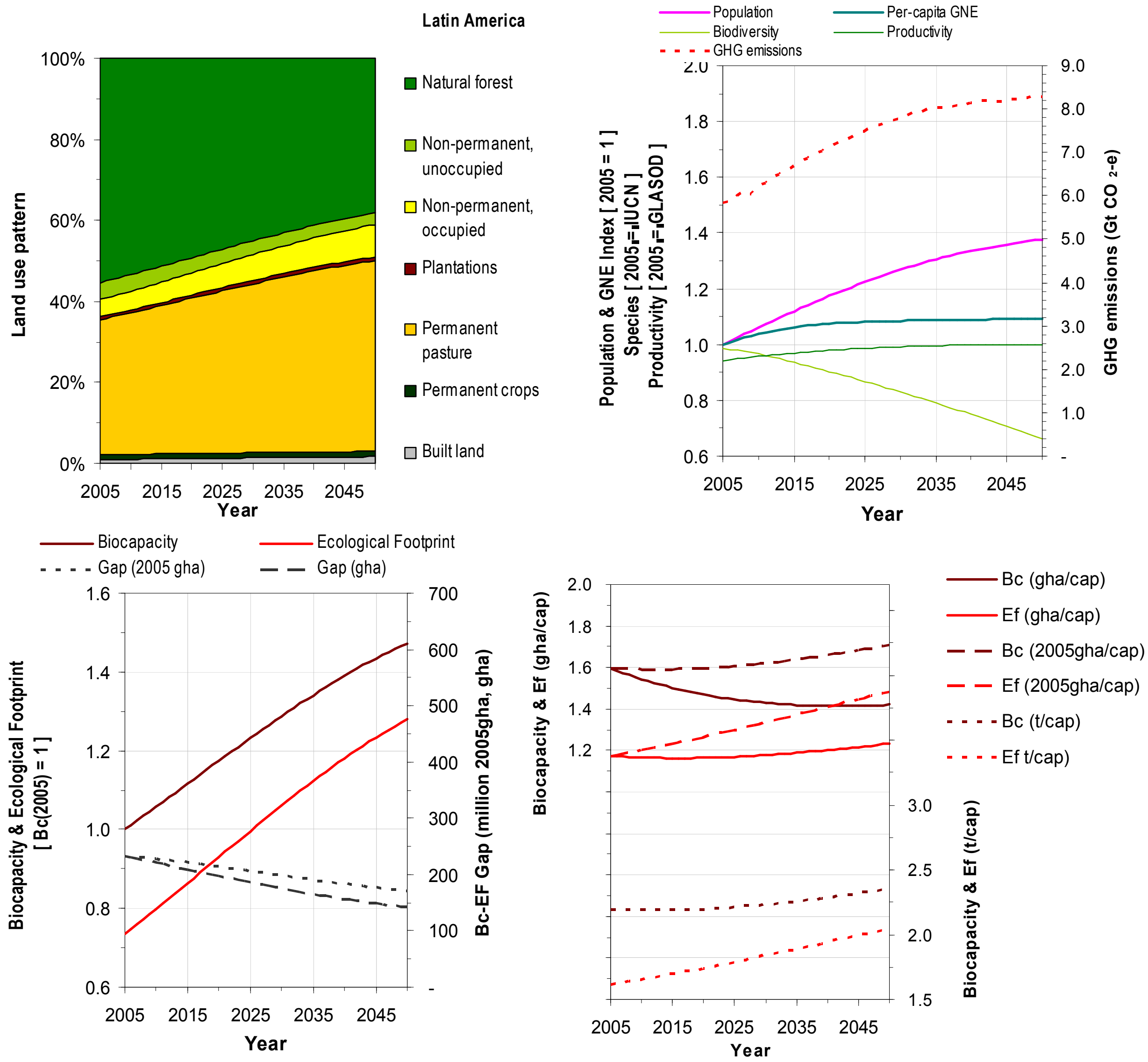

Fig. 17a: Future change in Latin America.

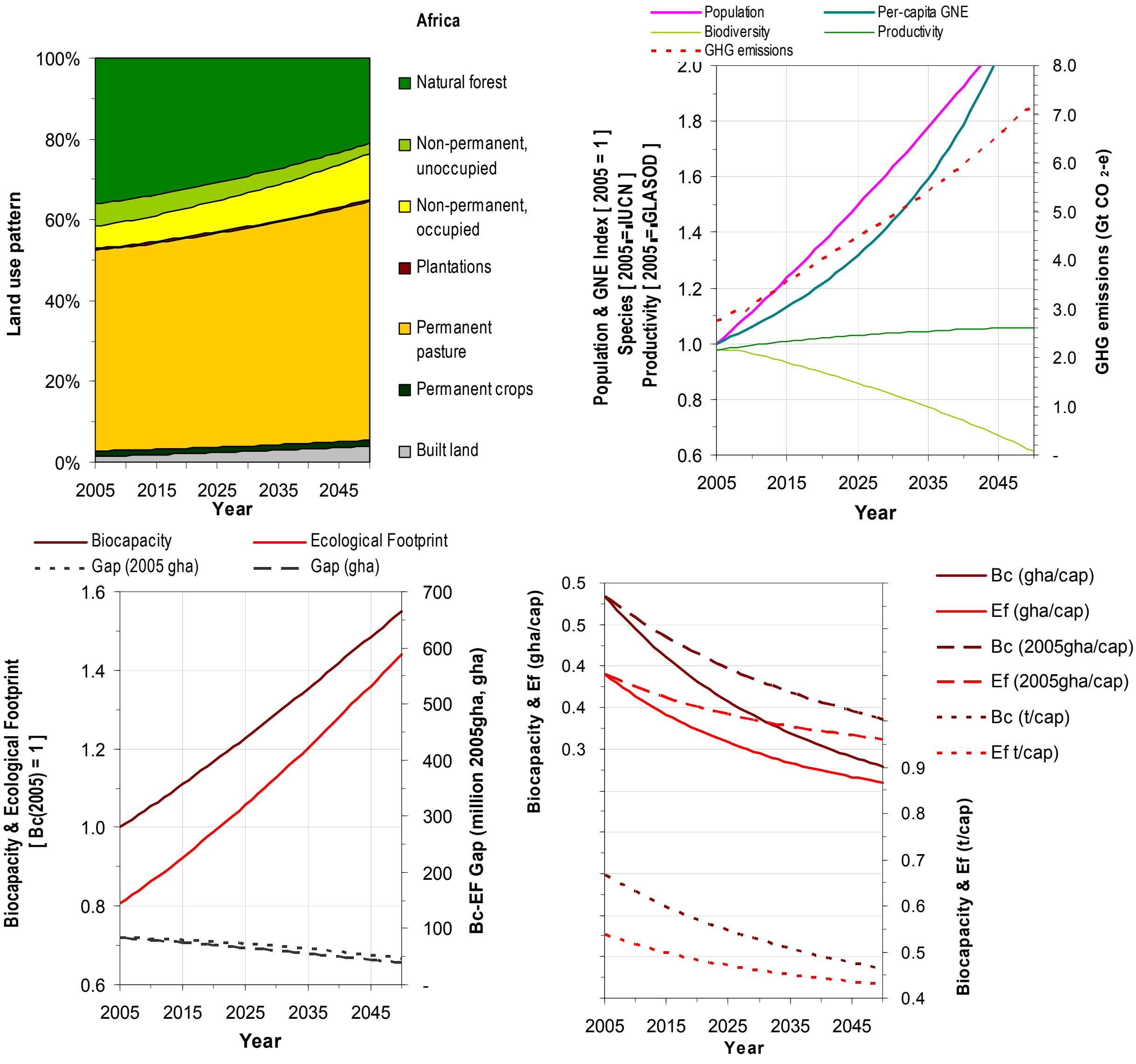

Fig. 17b: Future change in Africa.

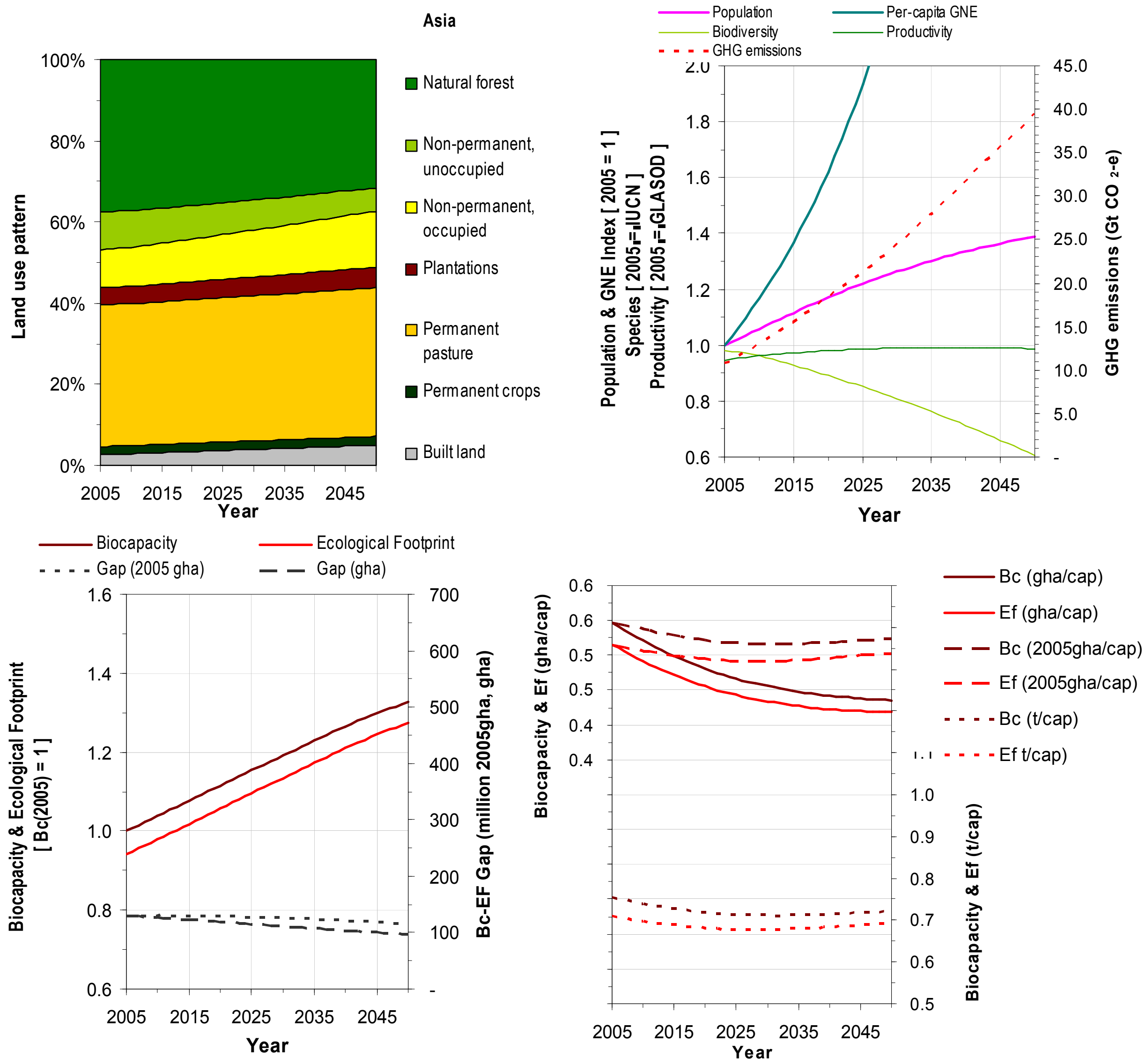

Fig. 17c: Future change in Asia.

Especially in Latin America and Africa the situation looks worrying: Supported by strong population growth, agricultural expansion is eating into natural systems at elevated rates, adding significant localised biodiversity threats to the overall threat from climate change. These trends cause an accelerated closure of the biocapacity-Ecological-Footprint gap, which in its magnitude outstrips the gap expansions on other continents.

It is interesting to compare these results with those obtained in a forecast by van Vuuren and Bouwman 2005. These authors use the IMAGE model for the spatial assessment of land use and cover, which includes CO_2 feedback on productivity, but excluding land degradation and biodiversity effects (paths 2, 4-12 and 14 in Fig. 1). It also includes world trade, however only "one layer deep" and not represented by a full input-output system. Van Vuuren and Bouwman 2005 show that under an A1-scenario[38], per-capita GDP grows between 2% and 4% annually (in agreement with Fig. 2), and the world per-capita Ecological Footprint decreases by about 0.2 gha (in agreement with Fig. 15). Van Vuuren and Bouwman 2005 report regional variations between 0.5 and 3 gha/cap for the land component, which agrees well with our results in Figs. 16 and 17.

In our analysis, permanent cropland and non-permanent arable land is set to increase by about 25% by 2050 in developing countries, and by less than 5% in developed countries. This finding is in good agreement with values projected for 2050 by Rosegrant and Ringler 1997 (p. 406) and Balmford *et al.* 2005. However, our analysis does not necessarily agree with other more detailed and spatially focused analyses currently in the literature: For Europe we estimate a decrease in agricultural land between 1% and 4%, and an increase in forest of above 1%. While the sign of these trends agrees with Rounsevell *et al.* 2005 and Rounsevell *et al.* 2006, the magnitude of changes in the latter authors' assessment is about a factor of 2-3 higher. For example, land use scenarios published for Europe as part of global change investigations (Ewert *et al.* 2005; Rounsevell *et al.* 2005) see larger reductions in agriculture that the current analyses we present. In particular, Rounsevell *et al.* 2006 see a range of 7-10% reductions each for crop and pastureland with increases of 5-8% increases in land that is surplus or used for biofuel production. On the other hand, van Vuuren and Bouwman 2005 see global land (as well as total land-based Ecological Footprint) under human use expand from 5.2 Gha in 2005 to almost 7 Gha in 2050, while our analysis concludes at slightly above 6 Gha in 2050.

Part of the difference is probably the semantics of classification, as land for biofuel could be used as intensively as crop- or pastureland particularly if it is given to switchgrass or Miscanthus production. As such it may pose a similar homogenisation threat to species biodiversity as does pasture for domestic animals, but probably less so than annually ploughed cropland. The key difference between the studies is the assumptions on the technological trajectory of crop yields. Ours are more guarded and assume a gradual plateauing of purely genetic gain around 2030 whereas those of Balmford *et al.* 2005 project linear extensions from the last 40 years to give a doubling of today's yields by 2050. Elsewhere we note our reticence on linear projections from the past because we question mankind's continuing ability to rely on pesticides and fertilisers, and whether cheap nitrogen fertiliser from natural gas will be as available after the mid 2020s when the oil peak may have passed, and the gas peak may be upon us. Additionally, Rounsevell *et al.* 2006 note in their discussion that the process of extensification and transition to organic agriculture is well underway in Europe because of people's concern about the health effects of intensive agriculture and its downstream effects on soil toxification and water pollution. A more extensive agriculture with less inputs might be kinder on biodiversity, but it might require even more land, or at least it might not save land.

Finally we estimate built land to increase by about 3 million hectares annually, which is much higher than $^1/_2$ million hectares annually estimated for urban land only by Rosegrant and Ringler 1997, but may be explained by additional built land in non-urban settlements, transport infrastructure, and mines.

4.3 Structural Decomposition Analysis

It is interesting to further query the closing of the biocapacity – Ecological Footprint gap (90 million gha (or 150 Mt) reduction between 2005 and 2050), by using Structural Decomposition Analysis. Our results (Fig. 18) support the projections in Fig. 15 by showing up population and affluence growth as major (negative) contributions to the reduction in the biocapacity – Ecological Footprint gap, contributing a reduction of about 50 million gha (or 80 Mt) each. As time proceeds, these reductions are increasingly exacerbated by biodiversity effects, which reach 70 million gha (120 Mt) by 2050. This is hardly surprising since "the current human-induced mass extinction may be of the same order of magnitude as the five other major extinction episodes which destroyed between 20 and 96 percent of existing species on the planet" (Gowdy 2000, p. 25).The combination of these closing forces cannot be offset by yield increases and the CO_2 fertilisation effect of climate change. If we assume gap closing rates of around 5 million gha per year (5-10 Mt/year, depending on yield) for the period following 2050, then it is unlikely that the world will be bioproductivity-constrained before the turn of the century (compare Rosegrant and Ringler 1997). However, by then, the gap would have shrunk to less than half of its 2005 size, and with the cessation of CO_2 fertilisation, biophysical limits could be reached within the first half of the 22nd century. Similarly, population and affluence growth are identified by van Vuuren and Bouwman 2005 as the main negative driving factors for the global Ecological Footprint.

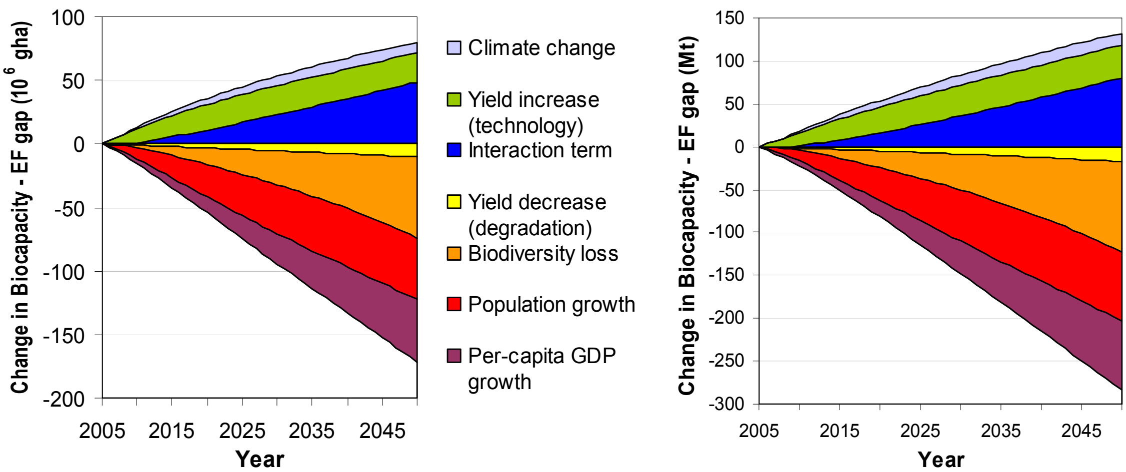

Fig. 18: Decomposition of the biocapacity-Ecological-Footprint gap trend in Fig. 15 (in millions of global hectares, left, or millions of tonnes, right).

The blue areas in Fig. 18 represent an interaction term, which covers the combined and synergistic effects of all variables (see Lenzen 2006a, De Bruyn 2000 Sec. 9.4, and Casler 2001 p.146–148). An example for a synergistic effect is the productivity loss due to the effect of global warming on biodiversity. In the Paasche method used to generate Fig. 18, this overall negative effect appears in both the light blue (climate change) and orange (biodiversity) areas. The interaction term must hence be positive in order to offset this double-counting.

4.4 Monte-Carlo simulation

Considering the significant uncertainties associated with some of the parameters in Eqs. 1–12, it is obvious that our results have to be qualified by a realistic assessment of error bounds. We first specified standard deviations for exogenous variables such as population and per-capita GNE and simulated 250 trajectories in 250 separate Monte-Carlo runs (Fig. 19 top two graphs), in order to reproduce future ranges specified in the literature. Then, in our Monte-Carlo simulation of Eqs. 1–12, these deviations propagate through to dependent variables such as productivity, land use, emissions, temperature, biodiversity, and the Ecological Footprint (Fig. 19 remaining graphs). With each subsequent modeling step, overall scenario uncertainty is influenced by two effects: First, it can increase as more uncertain parameters are included. Second, it can decrease because opposing stochastic perturbations in a sum over many countries cancel each other out. This is reflected in Fig. 19 by the differently shaped “cones”, or “fans”.

A mere reporting of numbers may instill a false sense of accuracy, and in this light, uncertainty assessment such as the one in this work are essential for decision-making. Identifying a preference for one of two or more policy options, for example, on the basis of their Ecological Footprint (endpoint) may be impossible on statistical grounds (failing hypothesis tests), however a multi-criteria comparison based on more certain mid-point indicators such as land use and greenhouse gas emissions may prove feasible (see Lenzen 2005).

In our Monte-Carlo analysis, population and per-capita GDP – the two exogenous variables in Eqs. 1-12 – are varied in a way to reproduce the variability published in the literature. Per-capita GDP is probably more volatile than the relatively inert world population, resulting in a wider cone. However, both cones are strictly increasing over time, and the year 2050 is likely to feature a population of 9 billion, generating on average around \$9,000PPP each. Future productivity is harder again to predict, with a host of underlying factors such as land degradation, the biodiversity-productivity relationship, fertilizer inputs, and future technology.

We have varied productivity parameters to an extent that allows both slight overall decreases and substantial (up to 50%) gains (compare Ewert *et al.* 2005; Rounsevell *et al.* 2005). Depending on the productivity outcome, land resources in form of natural forests and grasslands may experience a slight recovery (in case of substantial productivity increases), but more likely a 5-10% decrease. Influenced by the combined forces of land conversion and climate change, about 20-40% of species are projected to be under threats by 2050. In unison, all processes are estimated to lead to a narrowing of the biocapacity-Ecological-Footprint gap from 600 billion gha to around 450 billion gha. The small number of trajectories forecasting an increase in the gap result only if productivity can increase by 50% or thereabouts.

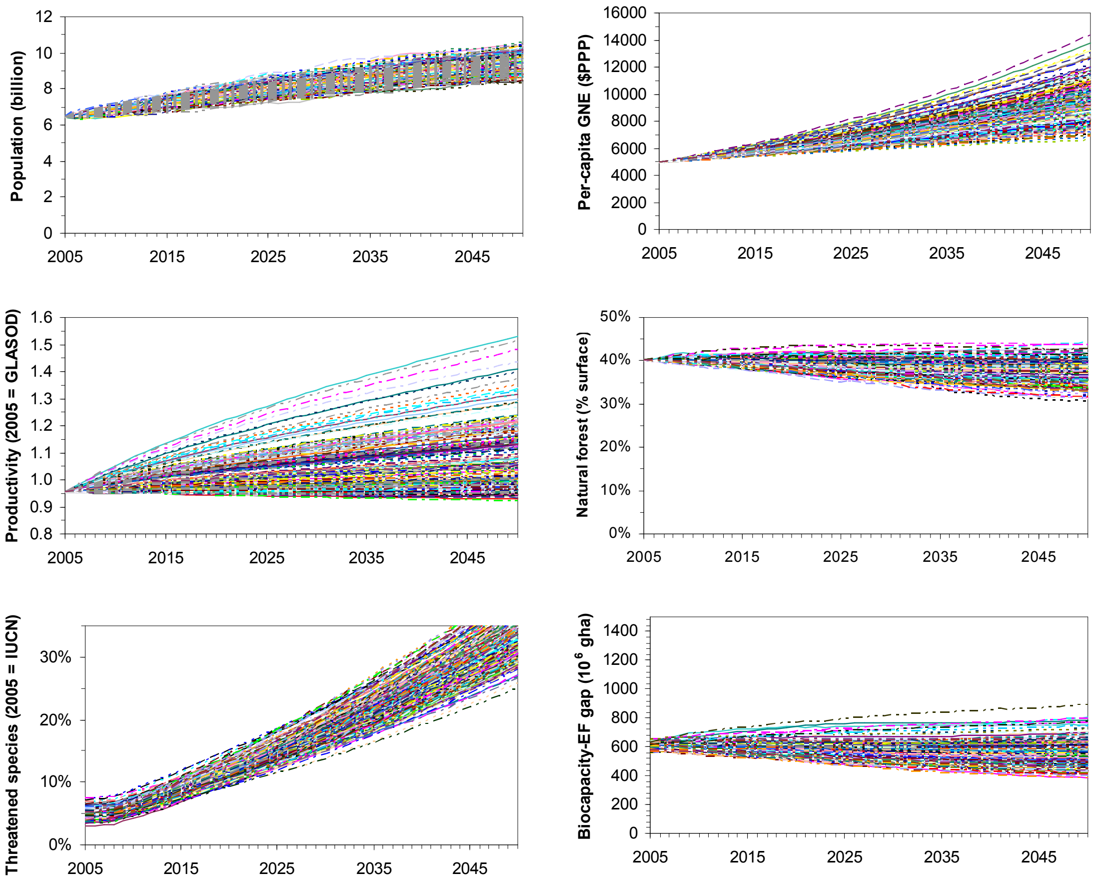

Fig. 19: Monte-Carlo analysis of uncertainties.

4.5 Corporate and other sub-national applications

It has always been the intention of Ecological Footprint practitioners to apply the concept to examine sub-national entities such as businesses (see for example Lenzen *et al.* 2003). In comparative assessments, static methods run into problems because they cannot capture the temporal profile of the causal chain in Fig. 1 (Lenzen *et al.* 2004b). Take for example two companies with identical greenhouse gas emission profiles over say 10 years. Assume that company 1 acted at the end of each financial year by planting trees to offset their annual carbon footprint, and that company 2 acted after 10 years by planting trees to offset the carbon footprint of the entire period. A static method would rank these companies as equally impacting after 10 years.

This ignores the fact that the greenhouse gases emitted by company 2 would have resided in the atmosphere for much longer than those of company 1, thus creating higher radiative forcing, and contributing more to climate change. Similar arguments apply to hypothetical situations of two companies, with one abating a short-lived impact, say land use and degradation, and with the other simultaneously abating a long-lived impact, say greenhouse gas emissions.[7] Finally, because of the vastly different atmospheric residence times of CO_2 and CH_4, the notion of land needed to sequester greenhouse gas emissions (as CO_2) is not applicable to CH_4 emissions unless a temporal method is applied.

The distortions described above can be overcome by applying a temporal analysis that is at least as long as the lifetime of the longest-living impact. It is straightforward to apply Eqs. 1-12 to companies.[45] Of particular benefit is the separability condition imposed during the derivation of Eq. 8 (see Meira and Miguez 2000), which enables partitioning contributions to temperature increases amongst emitters. In fact, the output of the entire system in Eqs. 1-12 is separable into mutually exclusive and collectively exhaustive pressure components.

We construct a hypothetical example of a power plant operator faced with a decision to twice (2008 and 2028) expand generating capacity because of increased demand. We assume that the operator has access to unused arable land and forest, and has the choice between two options: a) to clear about 16 hectares of land for a renewable energy array, and b) to add boiler and turbine capacity and step up fossil fuel combustion.

We investigate two cases: 1) renewables in 2008 and fossil expansion in 2028, and 2) fossil expansion in 2008 and renewables in 2028 (Fig. 20). After the second capacity expansion, both cases are nominally identical from a static point of view, that is a static Ecological Footprint method would not find any preference between the two decisions. In the following we will show that the temporal profiles of the two case studies matter. We will ignore any efficiency improvements, but we do assume spatial autocorrelation of species threats within and outside the plant operator's premises, by assigning the plant operator 100 hectares of land located between Belgium, Germany and Luxembourg, hosting 10 species.

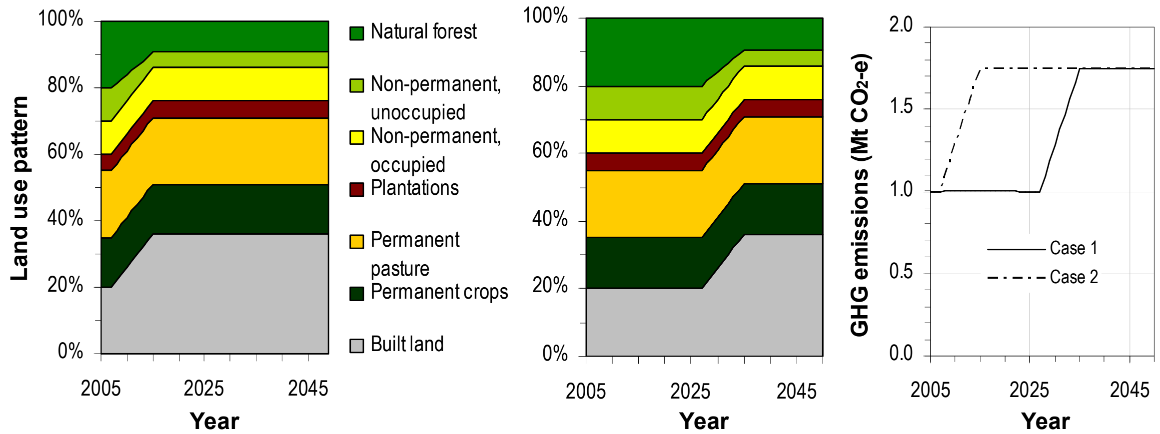

Fig. 20: Two case studies of a hypothetical expansion of land use and increase of annual greenhouse gas emissions, with different temporal profiles. In case 1, land clearing for a renewable array precedes a fossil-fuelled expansion, and in case 2 the sequence is reversed. Land use patterns: case 1 left graph, case 2 middle graph; 100% = 100 ha.

The temporal analysis shows that if renewables were introduced first, the clearing of forest and conversion of arable land to unproductive built land would cause an initial 1.8-gha (2.5-tonne) decrease of biocapacity on the site of the plant operator (Fig. 21, left graphs, blue area), with 0.4-gha (0.6-tonne) spill-over effects into neighbouring areas (mainly Germany

[45] This is with the exception of biocapacity, which may not be applicable to businesses.

and Belgium, purple). This would be accompanied by a slow underlying negative contribution (increased footprint) due to global warming from the emissions from land clearing (yellow). Off-site contributions represent effects on all other regions except the plant site. The on-site Ecological Footprint is zero, because we have assumed built land as having zero yield, but replacing productive land, which shows up as reduced biopcapacity. Note that decreases in the Ecological Footprint are shown as a positive contribution, and decreases in biocapacity as a negative contribution. The biocapacity trends turn steeper (more rapid decrease of biocapacity), and the emissions trend is just about to turn positive (decrease of the Ecological Footprint) after 2045, which is due to the delayed biodiversity effect from emissions commencing in 2028. In 2050, case 1 leads to a closing of the global biocapacity-Ecological-Footprint gap of about 3 gha (4 tonnes).

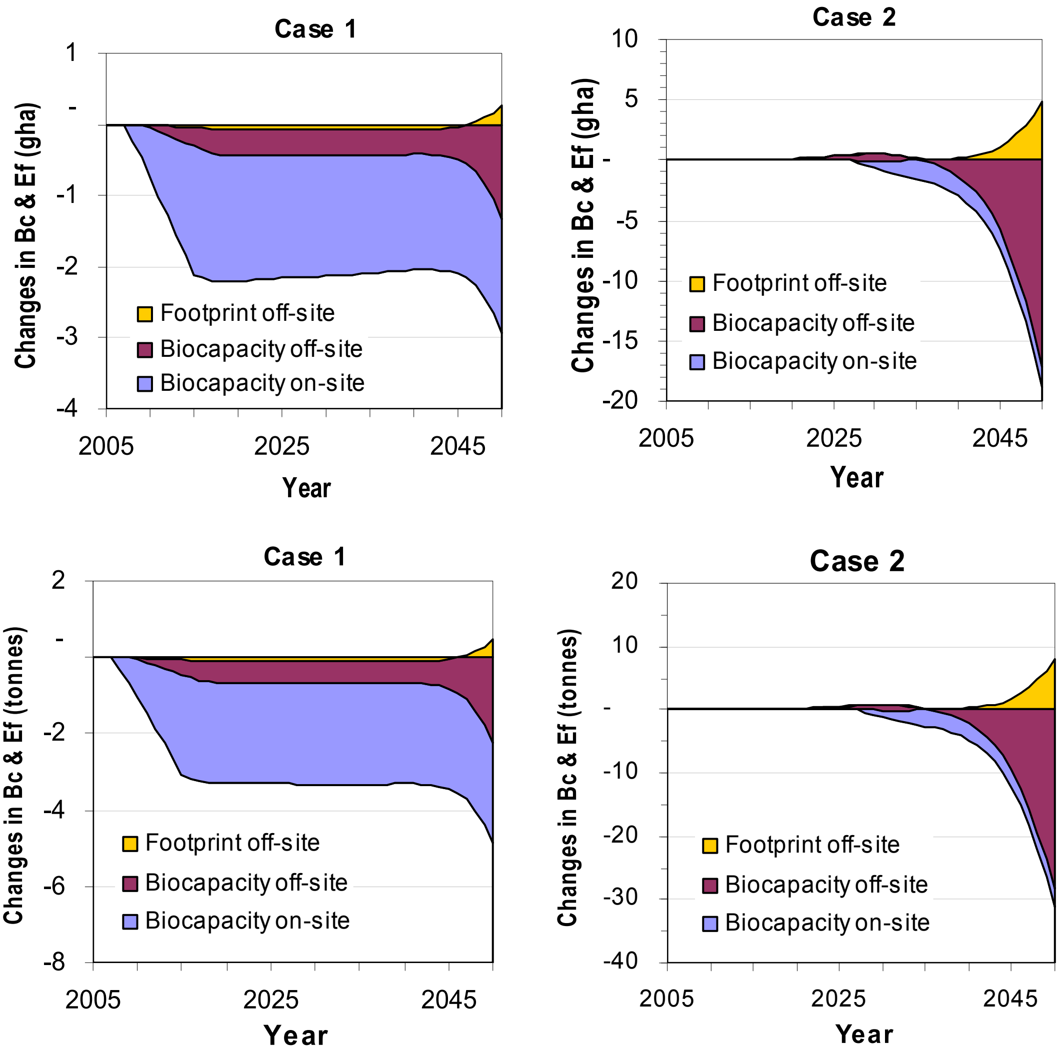

Fig. 21: Effect of case studies 1 (left) and 2 (right) on biocapacity and the Ecological Footprint, in units of global hectares (top) and tonnes (bottom).

Comparing the two left graphs in Fig. 21 shows the effect of using actual global hectares instead of tonnes. As the global yield $\overline{p}$ (see Eqs. 10 and 11) increases due to technology and climate, biocapacity and the Ecological Footprint decrease (top graph). This is different when using mass units because these are not scaled with global-average yield. Using constant, say 2005, global hectares (Global Footprint Network 2006) would show the same profiles as using tonnes.

Case 2 is markedly different: First, one can discern the on- and off-site land clearing effects from 2028 onwards, leading to a 1.8-gha (2.5-tonne) decrease of bioproductivity, just as in 2008 in case 1. About at the same time, the case 2 emissions from 2008 onwards start to show their delayed effect on biocapacity and the Ecological Footprint (both decreasing because of falling global yields). Case 2 terminates with a 2050 gap closing of 15 gha (about 22 tonnes).

Both decisions would be equivalent only after all effects of both the land clearing and emissions step changes in 2008 and 2028 have subsided. Considering the lifetimes of CO_2 fractions, this would take several centuries.

4.6 National Accounts, trade balances and Structural Path Analyses

In the following two Sub-sections, we will take two "cuts" at the causal chain network in Fig. 1: The first is at the pressure level, examining country rankings in terms of 2005 threatened species. The second is at the end-point level, examining the Ecological Footprint projected for 2050. By applying the input-output identity $\mathbf{E}^{(c)} = \mathbf{E}^{(p)}\hat{\mathbf{x}}^{-1}(\mathbf{I}-\mathbf{A})^{-1}\mathbf{y}$ of Ecological Footprints **E** projected for 2050 to the direct requirements matrix **A** of the world economy, the financial accounts of individual countries can be 'converted' into physical accounts. For example, in a production account the Ecological Footprint in any one country stands for (or is the result of) the total economic activity, or GDP, of this country. In a similar way, imports and exports can then be assigned an 'embodied' Ecological Footprint, in very much the same way as 'embodied' emissions can be assigned to financial transactions (Foran *et al.* 2005). Adding the trade balance to GDP results in the Ecological Footprint associated with the total consumption of a country (GNE).

4.6.1 2050 Ecological Footprint

Tab. 3 presents an example for such an Ecological Footprint account for 10 countries. Thus, production in the Côte d'Ivoire (a large producer of coffee, cocoa beans, and palm oil) directly appropriates 8 million global hectares. However its trading activities means that it affects 1 million gha elsewhere (imports) and has 2 million gha appropriated domestically by trading interactions with other countries (exports). The overall outcome is that domestic consumption requires 7 million gha, with 1 million gha of net exports, making the Côte d'Ivoire a net footprint exporter. By comparison, China's domestic economy appropriates 269 million gha (GNE), 255 in its own territories, 42 in other countries, it loses some of them by exporting 28, and thus shifts net 14 million gha of Ecological Footprint into other countries.

Country	*GDP*	+ Imports	− Exports	= *GNE*	*ΔT*
Côte d'Ivoire	8	1	2	7	1
Cambodia	11	0	1	10	1
Cameroon	10	0	1	9	1
Canada	154	17	41	129	25
Cape Verde	0	0	0	0	-0
Cayman Islands	-	0	-	0	-0
Central African Republic	4	0	0	4	0
Chad	2	0	0	2	0
Chile	12	2	3	12	0
China	255	42	28	269	-14

Tab. 3: Example 2050 National Ecological Footprint Accounts (million global hectares).

A ranked input-output analysis of the world economy (represented with one sector for each national economy) in terms of the 2050 Ecological Footprint yields clear and interesting results (Tab. 4): Large and/or wealthy and/or productive countries are the world's major appropriators of bioproductivity, simply because of their population size (China, India), the per-capita consumption (USA, Canada), and the productivity of local lands for exports (Malaysia, Brazil, Indonesia), respectively. While only affluent and/or populous nations (such as the USA Germany, China and Japan) import some of the largest Ecological Footprints into their borders, the list of exporters is joined by less affluent nations such as Brazil, Malaysia and Indonesia. The most striking result are probably the net trade balances: Developing countries in tropical regions – many of which contain biodiversity hotspots – use up their natural capital for the sake of providing exports destined for the developed world. This analysis underpins previous findings (Wackernagel 2000) about how nations can "live beyond their ecological means" by importing biocapacity from abroad.

Rank	GDP		Imports		Exports		GNE		Trade balance, top 10		Trade balance, bottom 10	
1	USA	484	USA	130	Russia	73	USA	581	Russia	65	USA	-98
2	Russia	370	Germany	42	Canada	41	Russia	305	Malaysia	32	Germany	-27
3	Brazil	260	China	42	Malaysia	38	China	269	Canada	25	Japan	-26
4	China	255	Japan	30	USA	32	Brazil	243	Brazil	17	United Kingdom	-24
5	Indonesia	163	United Kingdom	27	China	28	India	157	Indonesia	13	Netherlands	-19
6	Canada	154	France	24	Brazil	22	Indonesia	150	Venezuela	10	Italy	-17
7	India	147	Italy	22	Indonesia	19	Canada	129	Nigeria	8	Singapore	-15
8	Malaysia	89	Netherlands	22	Germany	16	France	77	Sweden	7	Belgium	-14
9	Mexico	68	Canada	17	Sweden	14	Germany	75	Congo	5	China	-14
10	France	66	Belgium	15	France	13	Mexico	67	Finland	5	Hong Kong	-13

Tab. 4: Ranking of countries in terms of the projected 2050 Ecological Footprint (million gha): territorial production (GDP), embodied in trade (imports, exports), embodied in consumption (GNE), as well as top and bottom 10 trade balances.

The Structural Path Analysis of the world's Ecological Footprint trade paints a similar picture: The top 30 Ecological Footprint trade flows are depicted in Table 5. Of the top 10 paths, 7 terminate with consumers in the USA, of these 7 paths, 6 originate in economies under development or in transition, and of those 6 paths, 5 originate in tropical or subtropical countries. Paths 10 to 30 mostly repeat this pattern, adding German, Chinese, Japanese, Dutch and Italian consumers, and Russia, Indonesia, Gabon, Congo and the USA as suppliers.

Rank	Trade flow	Path
1	53.0	Canada > United States of America
2	15.2	Mexico > United States of America
3	11.7	Malaysia > United States of America
4	10.8	China > United States of America
5	9.0	Venezuela > United States of America
6	8.5	United States of America > Canada
7	8.2	Nigeria > United States of America
8	7.7	Russian Federation > Areas, nes > India
9	7.3	Russian Federation > Germany
10	6.7	Brazil > United States of America
11	6.5	Malaysia > China
12	5.8	Russian Federation > United States of America
13	5.5	Russian Federation > China
14	5.4	United States of America > Mexico
15	5.4	Indonesia > Japan
16	5.0	Russian Federation > Netherlands
17	4.6	Russian Federation > Italy
18	4.5	Malaysia > Japan
19	4.5	Russian Federation > Ukraine
20	4.5	Russian Federation > Turkey
21	4.1	China > Japan
22	3.7	Indonesia > United States of America
23	3.6	Honduras > United States of America
24	3.5	Gabon > United States of America
25	3.4	Congo > China
26	3.2	Russian Federation > Belarus
27	3.1	Russian Federation > France
28	3.1	United States of America > Japan
29	3.0	Russian Federation > Poland
30	2.9	Russian Federation > United Kingdom

Tab. 5: Structural Path Analysis of Ecological Footprints embodied in world trade. The paths are to be interpreted as (example for Rank 3) "11.7 million gha are appropriated in Malaysia due to the consumption of goods and services in the USA". 'nes' = not elsewhere specified.

Note that path 8 is of 3rd order, representing 7.7 million gha supplied by Russia to India, but via other areas (not specified in the UN TradeCom statistics).

4.6.2 2050 Biocapacity – Ecological Footprint gap

An input-output analysis of the Biocapacity-Ecological Footprint gap shows that large and/or tropical developing countries enjoy the highest gaps between Ecological Footprint and available bioproductivity, and therefore the lowest risk (Tab. 6). Many of these developing countries use their remainders to export agricultural output to developed countries facing land constraints (narrow gaps), mostly within Europe.

Accounts of the rate of closure of the Biocapacity-Ecological Footprint gap (Tabs. 7 and 8) can be interpreted for example as follows: "Brazil's domestic production causes its biocapacity-Ecological-Footprint gap to close by 52,000 gha per year. Of these, 4,000 gha/year are destined for exports. Brazil has no significant imports from gap-closing countries, so that Brazil is a net exporter of 4,000 gha/year." These rates of closure were evaluated in units of 2005gha/year, in order to net out global yield increases. Tab. 7 indicates potential sustainability issues: Many developing countries show rapidly narrowing gaps, often used for exports of agricultural output. These developing nations increase their sustainability risk for the sake of exporting to developed nations, who use these imports to substitute environmentally intensive agricultural production.

A selected number of countries such as many European nations (for example because of rehabilitation of natural forests, and substitution of domestic agriculture with imports) and some former Soviet Republics (for example because of shrinking per-capita GDP) enjoy widening gaps, and export these to other (importing), often European or former Soviet states (Tab. 8). Finally, the USA, Canada and other large or industrious nations lead the chart of nations of net importers from countries with significant gap closure.

Rank	GDP		Imports		Exports		GNE		Trade balance, top 10		Trade balance, bottom 10	
1	Brazil	52	USA	15	Canada	11	Brazil	48	Canada	10	USA	-14
2	Russia	43	Germany	3	Russia	9	Russia	35	Russia	8	Germany	-3
3	Canada	40	China	3	Brazil	4	Canada	31	Brazil	4	China	-3
4	USA	12	United Kingdom	2	Sweden	3	USA	26	Sweden	3	United Kingdom	-2
5	Bolivia	10	France	2	Venezuela	2	Bolivia	9	Venezuela	1	Netherlands	-2
6	Sweden	8	Netherlands	2	Gabon	1	Australia	7	Gabon	1	Japan	-2
7	Australia	8	Italy	2	Australia	1	Congo (Zaïre)	7	Bolivia	1	France	-2
8	Congo (Zaïre)	7	Japan	2	Bolivia	1	Guyana	6	Guyana	1	Areas, nes	-1
9	Guyana	7	Areas, nes	1	Guyana	1	China	6	Australia	1	Italy	-1
10	Peru	4	Belgium	1	Congo	1	Sweden	6	Congo	1	India	-1

Tab. 6: Ranking of countries in terms of the Biocapacity-Ecological-Footprint gap (million gha) projected for 2050: territorial production (GDP), embodied in trade (imports, exports), embodied in consumption (GNE), as well as top and bottom 10 trade balances.

Rank	GDP		Imports		Exports		GNE		Trade balance, top 10	
1	Brazil	342	USA	44	Venezuela	29	Brazil	320	Venezuela	27
2	Bolivia	148	China	15	Brazil	29	Congo (Zaïre)	134	Brazil	22
3	Congo (Zaïre)	138	Japan	8	Gabon	19	Bolivia	132	Gabon	18
4	Venezuela	64	Brazil	7	Bolivia	16	USA	58	Bolivia	16
5	Australia	50	Indonesia	5	Malaysia	8	Indonesia	47	Australia	6
6	Indonesia	47	Korea, South	4	Australia	8	Australia	44	Malaysia	6
7	Myanmar	45	France	4	Indonesia	6	Myanmar	43	Ecuador	5
8	Gabon	42	Argentina	4	Ecuador	5	China	42	Congo (Zaïre)	5
9	Sudan	41	LAIA, nes	4	Congo (Zaïre)	5	Sudan	39	Chad	3
10	China	31	Germany	4	China	3	Venezuela	36	Sudan	3

Tab. 7: Ranking of countries in terms of the Biocapacity-Ecological-Footprint gap closure (‘000 2005gha/year) projected for 2050: top 10 in territorial production (GDP), embodied in trade (imports, exports), embodied in consumption (GNE), as well as trade balances.

Rank	GDP		Imports		Exports		GNE		Trade balance, bottom 10	
1	Russia	-72	Belarus	-0.38	Russia	-14	Russia	-56	USA	-43
2	Canada	-52	Kazakhstan	-0.22	Canada	-14	Canada	-35	Canada	-17
3	Sweden	-11	Norway	-0.18	Sweden	-4	Guyana	-9	Russia	-15
4	Guyana	-11	Lithuania	-0.18	Guyana	-2	Sweden	-7	China	-11
5	Namibia	-7	Cyprus	-0.16	Namibia	-1	Namibia	-6	Japan	-8
6	Ukraine	-5	Virgin Islands	-0.09	Norway	-1	Ukraine	-4	Sweden	-5
7	Georgia	-4	Denmark	-0.06	Suriname	-1	Georgia	-4	Korea, South	-4
8	Italy	-3	Latvia	-0.06	Italy	-1	Suriname	-2	France	-4
9	Suriname	-3	Hungary	-0.04	Ukraine	0	Serbia and Montenegro	-1	LAIA, nes	-4
10	Norway	-1	Slovakia	-0.04	Georgia	0	Norway	-1	Argentina	-4

Tab. 8: Ranking of countries in terms of the Biocapacity-Ecological-Footprint gap closure (‘000 2005gha/year) projected for 2050: bottom 10 in territorial production (GDP), embodied in trade (imports, exports), embodied in consumption (GNE), as well as trade balances.

The Structural Path Analysis of the gap closure rates (Tab. 9) confirms the National Accounts analyses: Almost all paths originate in heavily forested tropical and/or developing countries, and terminate in developed countries, thus lending more evidence to a bioproductivity displacement, or global "Footprint leakage" hypothesis.

The Unites States occupies 7 of the top 11 paths as a consumer of imports from gap-closing countries such as Venezuela, Gabon and Brazil. Other significant recipients are China and Japan, with imports originating also in a range of smaller tropical nations such as Ecuador, Congo, Malaysia, and Honduras.

Rank	Trade flow	Path
1	41.0	Canada > United States of America
2	32.1	Venezuela > United States of America
3	27.6	Gabon > United States of America
4	16.6	Brazil > United States of America
5	14.5	Bolivia > Brazil
6	8.4	Venezuela > LAIA, nes > Indonesia
7	6.6	Brazil > Argentina
8	6.0	Brazil > China
9	5.2	Mexico > United States of America
10	5.2	Bolivia > United States of America
11	5.1	Ecuador > United States of America
12	4.9	Venezuela > Areas, nes > India
13	4.4	Australia > Japan
14	4.1	Chad > United States of America
15	3.7	Bolivia > Argentina
16	3.4	Brazil > Germany
17	3.3	Congo > China
18	3.3	Sudan > China
19	3.1	Gabon > China
20	3.1	Malaysia > United States of America
21	3.0	Australia > China
22	2.9	Brazil > Mexico
23	2.6	Honduras > United States of America
24	2.6	Congo > United States of America
25	2.6	Brazil > Japan
26	2.4	Bolivia > Colombia
27	2.2	Russian Federation > Areas, nes > India
28	2.2	China > United States of America
29	2.2	Gabon > France
30	2.2	Indonesia > Japan

Tab. 9: Structural Path Analysis of sustainability in terms of gap closure rates caused by world trade. The paths are to be interpreted as (example for Rank 4) "Brazil closes its Biocapacity-Ecological-Footprint gap at a rate of 16,600 gha/year due to the consumption of goods and services in the USA". "nes" = not elsewhere specified.

5 Discussion

It must be emphasised that the present study – while aiming to be factually as accurate and realistic as possible in such a pilot study – is really about providing a blueprint for a method. Needless to say, the quantitative results presented in the previous Sections are preliminary, mainly because:

- We have not considered at all processes at a finer spatial detail. For example we use global averages for impacts of temperature increase on bioproductivity and biodiversity, whereas in reality these impacts differ widely on regional and local scales, often in different directions.
- We have ignored social trends such as effects on population growth of entrance of women into the workforce and onto electoral rolls. In our GDP/capita variable, we have not distinguished between benefitial and harmful components of GDP, such as in Genuine Progress Indicators (Hamilton 1999).
- As a global average, agricultural productivity is predicted to increase with a temperature increase of 1-3°C, and decrease above 3°C. As our forecast ends in 2050, and the temperature anomaly is below 2°C, we assume a positive relationship between temperature and productivity throughout.
- It is doubtful that results from country-scale analyses can be transferred to local scales, and vice versa. For example, experiments linking biodiversity with land use and bioproductivity are almost exclusively conducted at the plot or field scale, and there is no solid indication for their validity at large scales (Mattison and Norris 2005, p. 612).
- 90% of the variance in the greenhouse gas content is unexplained by GDP per capita (Fig. 12). This work could greatly benefit from having country-specific energy use and carbon intensity forecast up to 2100 instead of using a regression.
- Our multi-region input-output analysis assumes 1-sector economies.
- Our temporal analysis assumes some parameters as fixed over time, which becomes increasingly questionable as the future time frame increases. For example, a constant trade pattern **A** leaves no opportunity to increase trade.
- We have not modeled any price or other effects that invariably occur once a production factor such as land or useful species becomes scarce, and that may slow down resource depletion.
- Threatened species are often underreported, not all taxa have been surveyed, and only a minor fraction of potentially existing species are described.
- We have not sourced any data on the occupation of non-permanent arable lands, but instead assumed a 50% occupation in 2005. We have also ignored the fact that some natural areas displaced in our model by crop or grazing land may in reality not be suitable for agriculture or forestry, because of its steep terrain, soil fertility etc (see Rosegrant and Ringler 1997 p. 407).
- We have not appraised all paths listed in Fig. 1, and Fig. 1 is not an exhaustive representation of the causal pathways linking human consumption and bioproductivity. For example, we have not dealt with impacts on marine ecosystems and fish stocks, which could be accomplished by adding a separate fisheries module to Eqs. 1-12. We have also ignored a range of greenhouse gas sources such as soil emissions, CFC and HFC used in industrial processes, fugitive emissions, rice growing, cement production, etc.
- We have not considered the role of precipitation changes for changes in productivity as a result of climate change.

- We have not considered some major uncertainty sources, such as uncertainties in climate models in predicting future temperature anomaly distributions.
- In our model, the reaction time of bioproductivity and biodiversity changes following land use changes is essentially the time step of our finite-difference scheme, which is one year. We have neither specified any real impact delay-[46], life- or recovery-times, nor have we allowed for non-linearities, irreversibilities[47], threshold or hysteresis effects[48], for example the case of biodiversity re-establishing itself after cessation of land use or during site restoration.
- We have not modeled any trade responses. As a result, some countries, especially small island states, simply "run out" of land before 2050.
- Some of the parameters of the temporal analysis are not well enough understood and measured, and thus associated with high or unknown, and non-stochastic uncertainties.

Obviously, this analysis would improve if a) GIS or other spatially detailed data were incorporated, b) a multi-region, *multi-sector* input-output analysis were employed (see Turner *et al.* 2007, c) more links were included and quantified in the causal network, d) existing critical links and parameters were better measured, and e) if more causal pathways were evaluated in a dynamic way. The last point implies constructing more feedback links, say from bioproductivity to land use (humans reacting to declining bioproductivity by appropriating even more land), or from bioproductivity to consumption (human populations becoming constrained because of declining agricultural output).

Another major uncertainty for agricultural land use and population growth relates to biofuel production for liquid transport fuels (see for example Rounsevell *et al.* 2006). Responding to global change concerns for developed countries will require that transport fuel cycles become essentially decarbonised. Expanding ethanol production, particularly in Brazil and the USA, requires increasing amounts of maize and sugarcane as feedstock, with Brazil particularly being viewed as having almost unlimited potential. If biofuels move quickly from using food crops, to second generation methods based on forage cellulose and wood, there may be advantages for biodiversity conservation and landscape reclamation that come from perennial land cover and less frequent land disturbance.

Finally, there are types of impacts contained in Fig. 1 that can neither be related to, nor expressed as bioproductivity declines. An example is posed by Venetoulis and Talberth 2006; Venetoulis and Talberth 2007 who remark that in the existing static method, any damage can be done to non-arable land without it being penalised in Ecological Footprint figures. Impacts to non-arable lands are likely to influence services to humans via other pathways, for example by providing genetic "banks" that may aid in developing drugs or other chemicals, or by serving as tourist destinations (Beattie and Ehrlich 2001; Heywood *et al.* 2007), resilience (Arrow *et al.* 1995), or evolutionary potential (Gowdy 2000). Moreover, some of the mid-point variables in Fig. 1 could be perceived as end-points in their own right. For example, some species may not play any role whatsoever in supporting services to humans, but humans may place existence, options or bequest values on these species (Portney 1994; Gowdy 2000). Such effects will be missed in any Ecological Footprint approach focusing just on bioproductivity as an end-point: Our Structural Decomposition Analysis shows that the positive fertilisation effect of global warming on productivity may initially and for some time

[46] See Tilman *et al.* 1994.
[47] See Gowdy 2000, p. 35.
[48] Compare with Eppink *et al.* 2004, Footnote 2.

well outstrip the negative effects through reduced biodiversity, but land use and warming effects on biodiversity as such are clearly devastating. This confirms that productivity objectives – and therefore the Ecological Footprint – are at odds with biodiversity objectives (Lenzen *et al.* 2007a), and the dynamic Ecological Footprint proposed here still has to solve this shortcoming in order to avoid being rather "un-ecological". One way to get around this problem is by reporting on a number of incommensurable variables, rather than only on one (see the optional aggregation in Life-Cycle Impact Assessment), by aggregating using monetisation (Friedrich 2004), or by aggregating into a unit-less score using Multi-Criteria Decision Analysis (MCDA) methods (Goedkoop and Spriemsma 2001; Hertwich and Hammitt 2001; Diakoulaki and Grafakos 2004).

6 Conclusions

This work has laid the foundation for designing a dynamic Ecological Footprint approach grounded in ecology. This approach connects some previous Ecological Footprint modifications that were anchored at different points of the causal network, such as land use (Bicknell *et al.* 1998; McDonald and Patterson 2003), land disturbance and species diversity (Lenzen and Murray 2001; 2003), and pollution (Peters *et al.* 2007) to the original Ecological Footprint method measuring bioproductivity (Rees 1992; Wackernagel *et al.* 2005), and has thus demonstrated an effective and elegant means for unifying a range of methodologies and objectives into on framework while retaining the research question and metric of the original approach.

In our preliminary analysis we have demonstrated the capability of a dynamic approach, by collating a wide range of global country-level data, and applying state-of-the-art analytical techniques such as multi-region input-output analysis, multiple regression of spatial lag models, temporal climate modeling, Monte-Carlo simulation, Structural Decomposition Analysis, and Structural Path Analysis. These techniques can be applied to any consuming entity, ranging from the household or company scale up to region, States, and countries. The dynamic approach outlined here is meant to complement the existing, static Ecological Footprint method in that it is aimed at answering the question of how today's human activities contribute, through their particular land use and emission patterns, via habitat and biodiversity loss, and impaired ecosystem functioning, to future bioproductivity decline.

We have demonstrated that land use and biodiversity are long-term drivers for delayed bioproductivity losses, just as greenhouse gas emissions are a long-term driver for delayed climate change. Thus, even though bioproductivity is the quantity that is ultimately of interest to humans, we have to manage land use and biodiversity today in order to avoid dangerous losses of bioproductivity in the future, just as we have started to manage greenhouse gas emissions today in order to avoid potentially dangerous future climate change. A dynamic, causal method thus opens up new policy areas amenable to Ecological Footprint analysis, such as the investigation of agricultural practices, land disturbance, or species threats. Static, end-point methods are not sensitive towards these pressures.

Our results show that the biocapacity-Ecological-Footprint gap is likely to close by about 20% from 2005 until 2050, and that with the cessation of CO_2 fertilisation further reductions are likely to occur at accelerated rates of 5 million global hectares per year. Policy has the opportunity to curb these certainly bleak perspectives, and a dynamic method such as demonstrated in this work is able to give guidance to long-term planning. Amongst the suite of policy levers are conservation efforts ensuring protected, diverse and healthy ecosystems, a second "green revolution" in agriculture, intensified energy efficiency gains and decarbonisation of energy use, or education programs addressing population issues in the developing world, and affluence issues in the developed world.

In this context, we promote the dynamic nature of our methodological innovation: The world is unlikely to run out of bioproductive capacity for human purposes by the turn of the century. However by then, unless corrective action is taken, climate change and biodiversity declines will be well underway at accelerated rates, thus potentially leading to run-away productivity declines. Until then, expanding bioproductivity for human purposes remains fundamentally at odds with biodiversity conservation, and it is the capability to recognise early-warning drivers of delayed adverse impacts that facilitates the capability of implementing timely action.

Finally, we make a few comments on ways to advance the groundwork laid here: Undertaking a comprehensive analysis of a system of pathways is not a new idea: precedents are for example

– the ExternE project of the European Commission and the US Department of Energy (Rabl and Spadaro 2005), which developed an impact path methodology for the estimation and comparison of the environmental externality cost of various fuel cycles for electricity generation;
– the research undertaken by the Life-Cycle Assessment (LCA) community, including an interesting mid-point/end-point debate (Udo de Haes *et al.* 1999; Bare *et al.* 2000; Hertwich and Hammitt 2001), and ISO Standards (Klüppel 1998; Lecouls 1999; Ryding 1999; Marsmann 2000);
– the Intergovernmental Panel of Climate Change (IPCC; Intergovernmental Panel on Climate Change 2007) gathering evidence from research teams around the world on the multiple and complex links within and between the climatic, ecological, and societal systems.

Amongst the three programs, the LCA movement has made some inroads in coming up with a systematic characterisation of land use for impact assessment purposes (pathways 5 and 9 in Fig. 1), but there is no mandate (such as in the Ecological Footprint) to follow this pathway through to bioproductivity, and contrast it with available resources. The IPCC is concerned with land use and bioproductivity only to the extent that they connect to greenhouse gas emissions and climate change, for example as drivers through land clearing (pathway 4), or as degradation and damages (pathways 6, 7 and 13). Thus, while the Ecological Footprint community can and should make use of insights gained by other programs, it does not at present, and should not duplicate their purpose.

One principle common to the three programs above is that of appraising scientific results only in conjunction with uncertainty analyses. Both ExternE volumes and IPCC Assessment Reports present either error margins or cone diagrams of likely ranges for variables attached to future scenarios. A dynamic forecasting capability for the Ecological Footprint should follow these conventions. It is a well known fact that along with the progression of nodes in a causal pathway analysis, from pressure to state variables, the relevance to humans of the contained information increases, but so does the uncertainty of the analysis (see Lenzen 2005, and references therein). Uncertainty appraisals fulfill the critical role of demonstrating potential worst or best outcomes, and thus remind decision-makers of potential risks to hedge. Even though end-point uncertainties have been large enough to prohibit quantitative decision-making, undertakings such as the ExternE and IPCC programmes have contributed invaluable insights to research and policy agendas (Krewitt 2002). An extended Ecological Footprint approach could fulfill very much the same function.

Finally, each of the three initiatives have been extremely large-scale in terms of funding and academic and policy involvement, with working groups dealing with separate issues of the overall goal. Despite monumental efforts, many of the complex real-world interactions could only be modeled using empirical, aggregate relationships (such as dose-response curves, see Rabl and Spadaro 1999, or the diversity and bioproductivity elasticities used in this work). This may on one hand explain why the present analysis can deliver only a blueprint of a framework yet to come, and on the other hand demonstrate the challenges associated with coming up with an approach that can provide decision-makers with the direly needed tools to judge the impact that today's policies are likely to have on tomorrow's resources, environment and society.

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References

Aarssen, L.W. (1997). High productivity in grassland ecosystems: effected by species diversity or productive species? *Oikos* 80, 183-184.

Aarssen, L.W. (2001). On correlations and causations between productivity and species richness in vegetation: predictions from habitat attributes. *Basic and Applied Ecology* 2, 105-114.

Albrecht, J., François D. and Schoors K. (2002). A Shapley decomposition of carbon emissions without residuals. *Energy Policy* 30, 727-736.

Allameh, A., Safamehr A., Mirhadi S.A., Shivazad M., Razzaghi-Abyaneh M. and Naderi A. (2005). Evaluation of biochemical and production parameters of broiler chicks fed ammonia treated aflatoxin contaminated maize grains. *Animal Feed Science and Technology* 122, 289-301.

Amthor, J.S. and Members of the Ecosystems Working Group (1998). *Terrestrial Ecosystem Responses to Global Change: a research strategy*. Oak Ridge National Laboratory, Oak Ridge, USA.

Ang, B.W. (1999). Is the energy intensity a less useful indicator than the carbon factor in the study of climate change? *Energy Policy* 27(15), 943-946.

Ang, B.W. (2004). Decomposition analysis for policymaking in energy: which is the preferred method? *Energy Policy* 32, 1131-1139.

Ang, B.W. and Liu N. (2006). A cross-country analysis of aggregate energy and carbon intensities. *Energy Policy* 34(15), 2398-2404.

Antón, A., Castells F. and Montero J.I. (2007). Land use indicators in life cycle assessment. Case study: The environmental impact of Mediterranean greenhouses. *Journal of Cleaner Production* 15, 432-438.

Araújo, M.B. and Rahbek C. (2006). How does climate change affect biodiversity? *Science* 313, 1396-1397.

Armsworth, P.R., Kendall B.E. and Davis F.W. (2004). An introduction to biodiversity concepts for environmental economists. *Resource and Energy Economics* 26, 115-136.

Arrow, K., Bolin B., Costanza R., Dasgupta P., Folke C., Holling C.S., Jansson B.-O., Levin S., Mäler K.-G., Perrings C. and Pimentel D. (1995). Economic growth, carrying capacity and the environment. *Science* 268(5210), 520-521.

Asafu-Adjaye, J. (2003). Biodiversity loss and economic growth: a cross-country analysis. *Contemporary Economic Policy* 21(2), 173-185.

Asner, G.P., Elmore A.J., Olander L.P., Martin R.E. and Harris A.T. (2004). Grazing systems, ecosystem responses, and global change. *Annual Review of Environment and Resources* 29, 261-299.

Auhagen, A. (1994). *Wissenschaftliche Grundlagen zur Berechnung einer Ausgleichsausgabe*. Im Auftrag der Senatsverwaltung für Stadtentwicklung und Umweltschutz Abt. III, Auhagen & Partner GmbH, Berlin, Germany.

Australian Bureau of Agricultural and Resource Economics (2004). *Australian Commodity Statistics*. ABARE project 1059, Australian Government, Canberra, Australia.

Australian Bureau of Statistics (2000). *1998-99 Household Expenditure Survey - Detailed Expenditure Items*. ABS Catalogue No. 6535.0, Australian Bureau of Statistics, Canberra, Australia.

Australian Bureau of Statistics (2004). *Australian National Accounts, Input-Output Tables, 1998-99*. ABS Catalogue No. 5209.0, Australian Bureau of Statistics, Canberra, Australia.

Ayres, R.U. (2000). Commentary on the utility of the ecological footprint concept. *Ecological Economics* 32, 347-349.

Baitz, M., Kreißig J. and Schöch C. (1998). *Method to integrate land use in Life Cycle Assessment*. IKP Universität Stuttgart, Abt. Ganzheitliche Bilanzierung, Stuttgart, Germany.

Balmford, A., Green R.E. and Scharlemann J.P.W. (2005). Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. *Global Change Biology* 11, 1594-1605.

Bare, J.C., Hofstetter P., Pennington D.W. and Udo de Haes H.A. (2000). Life cycle impact assessment workshop summary. Midpoints versus endpoints: the sacrifice and benefits. *International Journal of Life Cycle Assessment* 5(6), 319-326.

Barthlott, W., Lauer W. and Placke A. (1996). Global distribution of species diversity in vascular plants: towards a world map of phytodiversity. *Erdkunde* 50(4), 317-327.

Bashmakov, I. (2007). Three laws of energy transition. *Energy Policy* 35(7), 3583-3594.

Beattie, A. and Ehrlich P.R. (2001). *Wild Solutions - How Biodiversity is Money in the Bank*. Melbourne University Press Melbourne, Australia.

Bicknell, K.B., Ball R.J., Cullen R. and Bigsby H.R. (1998). New methodology for the ecological footprint with an application to the New Zealand economy. *Ecological Economics* 27(2), 149-160.

Blasing, T.J. and Smith K. (2006). *Recent greenhouse gas concentrations*. Internet site http://cdiac.ornl.gov/pns/current_ghg.html, Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, USA.

Blonk, H., Lindeijer E. and Broers J. (1997). Towards a methodology for taking physical degradation of ecosystems into account in LCA. *International Journal of Life Cycle Assessment* 2(2), 91-98.

Boisvenue, C. and Running S.W. (2006). Impacts of climate change on natural forest productivity - evidence since the middle of the 20th century. *Global Change Biology* 12, 862-882.

Bradford, G.E. (1999). Contributions of animal agriculture to meeting global food demand. *Livestock Production Science* 59, 95-112.

Bradley, M. and Peter D. (1998). *An LCA of the production of a daily newspaper and a weekly magazine.*, Axel Springer Verlag, STORA and CANFOR, INFRAS, Vancouver, Canada.

Brentrup, F. and Küsters J. (2002). Life Cycle Impact Assessment of land use based on the hemeroby concept. *International Journal of Life Cycle Assessment* 7(6), 339-348.

Browne, J. and Heske E. (1990). Control of a desert-grassland transition by a keystone rodent guild. *Science* 250, 1705-1707.

Bullard, C.W. and Sebald A.V. (1977). Effects of parametric uncertainty and technological change on input-output models. *Review of Economics and Statistics* LIX, 75-81.

Bullard, C.W. and Sebald A.V. (1988). Monte Carlo sensitivity analysis of input-output models. *The Review of Economics and Statistics* LXX(4), 708-712.

Casler, S.D. (2001). Interaction terms and structural decomposition: an application to the defense cost of oil. In: Lahr, M.L. and Dietzenbacher, E., Eds. *Input-Output Analysis: Frontiers and Extensions*. Palgrave MacMillan, London, UK, 143-159.

Central Intelligence Agency (2006). *The World Factbook*. Internet site https://www.cia.gov/cia/publications/factbook/geos/, Central Intelligence Agency (CIA), Washington D.C., USA.

Chapin, F.S., Walker B.H., Hobbs R.J., Hooper D.U., Lawton J.H., Sala O.E. and Tilman D. (1997). Biotic control over the functioning of ecosystems. *Science* 277, 500-504.

Chapin, F.S., Zavaleta E.S., Ebiner V.T., Naylor R.L., Vitousek P.M., Reynolds H.L., Hooper D.U., Lavorel S., Sala O.E., Hobbie S.E., Mack M.C. and Diaz S. (2000). Consequences of changing biodiversity. *Nature* 405, 234-242.

Chen, B. and Chen G.Q. (2006). Ecological footprint accounting based on emergy - A case study of the Chinese society. *Ecological Modelling* 198, 101-111.

Chen, B. and Chen G.Q. (2007). Modified ecological footprint accounting and analysis based on embodied exergy - a case study of the Chinese society 1981-2001. *Ecological Economics* 61, 355-376.

Christenesen, N.L., Bartuska A.M., Brown J.H., Carpenter S., D'Antonio C.D., Francis R., Franklin J.F., MacMahon J.A., Noss R.F., Parsons D.J., Peterson C.H., Turner M.G. and Woodmansee R.G. (1996). The report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. *Ecological Applications* 6, 665-691.

Cliff, A. and Ord J. (1981). *Spatial Processes, Models and Applications*. Pion London, UK.

Cole, M.A., Rayner A.J. and Bates J.M. (1997). The environmental Kuznets curve: an empirical analysis. *Environment and Development Economics* 2(4), 401-416.

Conway, G. and Toenniessen G. (1999). Feeding the world in the twenty-first century. *Nature* 402, C55-C58.

Costanza, R. (2000). The dynamics of the ecological footprint concept. *Ecological Economics* 32, 341-345.

Costanza, R., Fisher B., Mulder K., Liu S. and C. T. (2007). Biodiversity and ecosystem srvices: A multi-scale empirical study of the relationship between species richness and net primary production. *Ecological Economics* 61, 478-491.

Cowell, S.J. (1998). *Environmental Life Cycle Assessment of agricultural systems: Integration into decision-making*. Ph.D. Dissertation, University of Surrey, Guildford, UK.

Crama, Y., Defourny J. and Gazon J. (1984). Structural decomposition of multipliers in input-output or social accounting matrix analysis. *Economie Appliquée* 37, 215-222.

Cramer, W., Bondeau A., Woodward F.I., Prentice I.C., Betts R.A., Brovkin V., Cox P.M., Fisher V., Foley J.A., Friend A.D., Kucharik C., Lomas M.R., Ramankutty N., Sitch S., Smith B., White A. and Young-Molling C. (2001). Global response of terrestrial ecosystem structure and function to CO_2 and climate change: results from six dynamic global vegetation models. *Global Change Biology* 7, 357-373.

Cubasch, U., Meehl G.A., Boer G.J., Stouffer R.J., Dix M., Noda A., Senior C.A., Raper S. and Yap K.S. (2001). Projections of Future Climate Change. In: Houghton, J.T., Ed. *Climate Change 2001: The Scientific Basis. Contribution of Working Group I to the Third Assessment Report of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, UK.

Daily, G., Ed. (1997). *Nature's services. Societal dependence on natural ecosystems*. Island Press, Washington, D C, USA.

de Bruyn, S.M. (2000). The Environmental Kuznets Curve hypothesis. In: de Bruyn, S.M., Ed. *Economic Growth and the Environment*. Kluwer Academic Publishers, Dordrecht, Netherlands, 77-98.

De Bruyn, S.M. (2000). Decomposition analysis: a tool to investigate the relationship between emissions and income. In: de Bruyn, S.M., Ed. *Economic Growth and the Environment*. Kluwer Academic Publishers, Dordrecht, Netherlands, 163-184.

Deacon, R.T. (1994). Deforestation and the rule of law in a cross-section of countries. *Land Economics* 70(4), 414-430.

Defourny, J. and Thorbecke E. (1984). Structural path analysis and multiplier decomposition within a social accounting matrix framework. *Economic Journal* 94, 111-136.

Delgado, C., Rosegrant M., Steinfeld H., Ehui S. and Courbois C. (2001). Livestock to 2020: the next food revolution. *Outlook On Agriculture* 30(1), 27-29.

Diakoulaki, D. and Grafakos S., Eds. (2004). *Multicriteria Analysis*. ExternE - Externalities of Energy: Extension of accounting framework and Policy Applications. Directorate-General for Research, Sustainable Energy Systems, Luxembourg-Ville, Luxembourg.

Dietzenbacher, E. and Los B. (1998). Structural decomposition techniques: sense and sensitivity. *Economic Systems Research* 10(4), 307-323.

Dormann, C.F. (2007). Promising the future? Global change projections of species distributions. *Basic and Applied Ecology*, in press.

Ehrhardt-Martinez, K., Crenshaw E.M. and Jenkins J.C. (2002). Deforestation and the Environmental Kuznets Curve: a cross-national investigation of intervening mechanisms. *Social Science Quarterly* 83(1), 226-243.

Ekins, P. (1997). The Kuznets curve for the environment and economic growth: examining the evidence. *Environment and Planning A* 29(5), 805-830.

Ekvall, T. (1998). Land use in LCA - existing characterization methods. In: *Cost Action E9 Workshop*, London, UK, Imperial College

Emmerson, M.C., Solan M., Emes C., Peterson D.M. and Raffaelli D. (2001). Consistent patterns and the idiosyncratic effects of biodiversity in marine ecosystems. *Nature* 411, 73-77.

Eppink, F.V., Van den Bergh J.C.J.M. and Rietveld P. (2004). Modelling biodiversity and land use: urban growth, agriculture and nature in a wetland area. *Ecological Economics* 51, 201-216.

Ewert, F., Porter J.R. and Rounsevell M.D.A. (2007). Crop models, CO_2, and climate change. *Science* 315, 459-460.

Ewert, F., Rounsevell M.D.A., Reginster I., Metzger M.J. and Leemans R. (2005). Future scenarios of European agricultural land use - I. Estimating changes in crop productivity. *Agriculture, Ecosystems and Environment* 107, 101-116.

FAO (2007). *Global Fibre Supply Model (GFSM)*. Internet site http://www.fao.org/docrep/006/x0105e/x0105e00.htm, Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO Statistics Division (FAOSTAT) (2001). *Forested Area*. Global Forest Resources Assessment 2000 — Main Report, http://www.fao.org/forestry/fo/fra/index.jsp, Food and Agriculture Organisation of the United Nations (FAO), Rome, Italy.

FAO Statistics Division (FAOSTAT) (2004). *Land Use - 2002*. FAOSTAT on-line statistical service http://apps.fao.org, Food and Agriculture Organisation of the United Nations (FAO), Rome, Italy.

FAO Statistics Division (FAOSTAT) (2007a). *Database on Macro-economic Indicators*. Internet site http://www.fao.org/statistics/os/macro_eco/default.asp, Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO Statistics Division (FAOSTAT) (2007b). *Harvested area and production quantity*. Internet site http://faostat.fao.org/site/336/default.aspx, Food and Agriculture Organization of the United Nations, Rome, Italy.

FAO Statistics Division (FAOSTAT) (2007c). *ForeSTAT*. Internet site http://faostat.fao.org/site/381/default.aspx, Food and Agriculture Organization of the United Nations, Rome, Italy.

Federative Republic of Brazil (1997). *Proposed elements of a protocol to the United Nations Framework Conventions on Climate Change, presented by Brazil in response to the Berlin Mandate*. Document FCCC/AGBM/1997/MISC.1/Add.3, Internet site http://www.mct.gov.br/clima/ingles/quioto/propbra.htm, Brasília, Brazil.

Felten, P. and Glod S. (1995). *Weiterentwicklung ökologischer Indikatoren für die Flächenbeanpruchung und für Lärmwirkungen und Anwendung auf Logistik-Konzepte einer Firma*. Semesterarbeit, ETH, Zürich, Switzerland.

Ferng, J.-J. (2001). Using composition of land multiplier to estimate ecological footprints associated with production activity. *Ecological Economics* 37, 159-172.

Ferng, J.-J. (2002). Toward a scenario analysis framework for energy footprints. *Ecological Economics* 40, 53-69.

Ferng, J.-J. (2005). Local sustainable yield and embodied resources in ecological footprint analysis - a case study on the required paddy field in Taiwan. *Ecological Economics* 53, 415-430.

Foran, B., Lenzen M. and Dey C. (2005). *Balancing Act - A Triple Bottom Line Account of the Australian Economy*. Internet site http://www.isa.org.usyd.edu.au, CSIRO Resource Futures and The University of Sydney, Canberra, ACT, Australia.

Foreign Agricultural Service (2004). *The Amazon: Brazil's final soybean frontier. Production Estimates and Crop Assessment Division*. Internet site http://www.fas.usda.gov/pecad2/highlights/2004/01/Amazon/Amazon_soybeans.htm, United States Department of Agriculture (USDA), Washington D.C., USA.

Friedrich, R., Ed. (2004). *New Ext - New Elements for the Assessment of External Costs from Energy Technologies, Final Report to the European Commission, DG Research, Technological Development and Demonstration (RTD)*. ExternE - Externalities of Energy. Directorate-General for Research, Sustainable Energy Systems, Luxembourg-Ville, Luxembourg.

Fung, I.Y., Doney S.C., Lindsay K. and John J. (2005). Evolution of carbon sinks in a changing climate. *Proceedings Of The National Academy Of Sciences Of The United States Of America* 102, 11201-11206.

Gao, C., Jiang D., Wang D. and Yan J. (2006). Calculation of ecological footprint based on modified method and quantitative analysis of its impact factors - A case study of Shanghai. *Chinese Geographical Science* 16, 306-313.

Geneletti, D. (2007). An approach based on spatial multicriteria analysis to map the nature conservation value of agricultural land. *Journal of Environmental Management* 83, 98-105.

Gitay, H., Suarez A., Watson R.T. and Dokken D.J., Eds. (2002). *Climate Change and Biodiversity - Intergovernmental Panel on Climate Change.*

Glanznig, A. (1995). *Native vegetation clearance, habitat loss and biodiversity decline*. Biodiversity Series, Paper No. 6, Department of the Environment, Sport and Territories Biodiversity Unit, Canberra, Australia.

Global Footprint Network (2006). *Constant Global Hectares*. Discussion Document for the National Accounts Committee, Global Footprint Network, Oakland, USA.

Global International Waters Assessment (2006). *Challenges to International Waters: Regional Assessments in a Global Perspective*. Internet site http://www.giwa.net/publications/finalreport/, United Nations Environment Program, Nairobi, Kenya.

Global Land Cover Facility (GLCF) (2002). *Modis Satellite Imagery: Forested Area*. MODIS 500m Vegetation Continuous Fields Percent Tree Cover, http://glcf.umiacs.umd.edu/data/, University of Maryland, Maryland, USA.

Goedkoop, M. and Spriemsma R. (2001). *The Eco-indicator 99 - a damage oriented method for Life Cycle Assessment*. Internet site http://www.pre.nl/eco-indicator99/ei99-reports.htm, PRé Consultants B.V., Amersfort, Netherlands.

Government Accountability Office (2007). *Crude Oil: Uncertainty about future oil supply makes it important to develop a strategy for addressing a peak and decline in oil production*. Report to Congressional Office Requesters GAO 07-293, Internet site http://www.gao.gov/new.items/d07283.pdf, United States Government Accountability Office, Washington D.C., USA.

Gowdy, J.M. (2000). The value of biodiversity: markets, society and ecosystems. *Land Economics* 73(1), 25-41.

Graus, W.H.J., Voogt M. and Worrell E. (2007). International comparison of energy efficiency of fossil power generation. *Energy Policy* 35(7), 3936-3951.

Grime, J.P. (1997). Biodiversity and ecosystem function; the debate deepens. *Science* 277, 1260-1261.

Grossman, G.M. and Krueger A.B. (1995). Economic growth and the environment. *Quarterly Journal of Economics* 110(2), 353-377.
Guisan, A. and Thuiller W. (2005). Predicting species distribution: offering more than simple habitat models. *Ecology Letters* 8(993-1009).
Guterman, L. (2000). Have ecologists oversold biodiversity? *Chronicle of Higher Education*(13 October).
Hamilton, C. (1999). The genuine progress indicator: methodological developments and results from Australia. *Ecological Economics* 30, 13-28.
Hampe, A. (2004). Bioclimate envelope models: what they detect and what they hide. *Global Ecology And Biogeography* 13, 469-471.
Hampicke, U. (1991). *Naturschutz-Ökonomie*. Ulmer Stuttgart, Germany.
Hansen, J.E., Ruedy R. and Lo K. (2006). *Global annual temperature anomalies (degrees C) computed from land and ocean data, 1880-2005, NASA GISS Surface Temperature (GISTEMP) Analysis*. Internet site http://cdiac.ornl.gov/trends/temp/hansen/hansen.html, Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, U.S.A.
Hector, A., Schmid B., Beierkuhnlein C., Caldeira M.C., Diemer M., Dimitrakopoulos P.G., Finn J.A., Freitas H., Giller P.S., Good J., Harris R., Högberg P., Huss-Danell K., Joshi J., Jumpponen A., Körner C., Leadley P.W., Loreau M., Minns A., Mulder C.P.H., O'Donovan G., Otway S.J., Pereira J.S., Prinz A., Read D.J., Scherer-Lorenzen M., Schulze E.-D., Siamantziouras A.-S.D., Spehn E., Terry A.C., Troumbis A.Y., Woodward F.I., Yachi S. and Lawton J.H. (2000). No consistent effect of plant diversity on productivity: Response. *Science* 289, 1255a.
Hector, A., Schmid B., Beierkuhnlein C., Caldeira M.C., Diemer M., Dimitrakopoulos P.G., Finn J.A., Freitas H., Giller P.S., Good J., Harris R., Hogberg P., Huss-Danell K., Joshi J., Jumpponen A., Korner C., Leadley P.W., Loreau M., Minns A., Mulder C.P.H., O'Donovan G., Otway S.J., Pereira J.S., Prinz A., Read D.J., Scherer-Lorenzen M., Schulze E.D., Siamantziouras A.S.D., Spehn E.M., Terry A.C., Troumbis A.Y., Woodward F.I., Yachi S. and Lawton J.H. (1999). Plant diversity and productivity experiments in European grasslands. *Science* 286, 1123-1127.
Heijungs, R., Guinée J. and Huppes G. (1997). *Impact categories for natural resources and land use*. CML Report 138, Centre of Environmental Science, Leiden, Netherlands.
Herendeen, R. (1978). Total energy cost of household consumption in Norway, 1973. *Energy* 3, 615-630.
Herendeen, R., Ford C. and Hannon B. (1981). Energy cost of living, 1972-1973. *Energy* 6, 1433-1450.
Herendeen, R. and Tanaka J. (1976). Energy cost of living. *Energy* 1, 165-178.
Hertwich, E.G. and Hammitt J.K. (2001). Decision-analytic framework for impact assessment, Part 2: Midpoints, endpoints and criteria for method development. *International Journal of Life Cycle Assessment* 6, 265-272.
Heywood, V., Casas A., Ford-Lloyd B., Kell S. and Maxted N. (2007). Conservation and sustainable use of crop wild relatives. *Agriculture, Ecosystems and Environment*(121), 245-255.
Hickling, R., Roy D.B., Hill J.K., Fox R. and Thomas C.D. (2006). The distributions of a wide range of taxonomic groups are expanding polewards. *Global Change Biology* 12, 450-455.
Hoekstra, R. and van den Bergh J.C.J.M. (2002). Structural decomposition analysis of physical flows in the economy. *Environmental and Resource Economics* 23, 357-378.
Hooper, D.U., Chapin F.S., Ewel J.J., Hector A., Inchausti P., Lavorel S., Lawton J.H., Lodge D.M., Loreau M., Naeem S., Schmid B., Setälä H., Symstad A.J., Vandermeer J. and Wardle D.A. (2005). Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecological Monographs* 75, 3-35.
Houghton, J.T., Meira Filho L.G., Callander B.A., Harris N., Kattenberg A. and Maskell K., Eds. (1996). *Climate Change 1995: The Science of Climate Change*. Intergovernmental Panel on Climate Change, Cambridge, UK.
Houghton, R.A. (2003). *Emissions (and Sinks) of Carbon from Land-Use Change (Estimates of national sources and sinks of carbon resulting from changes in land use, 1950 to 2000)*. Report to the World Resources Institute, Woods Hole Research Center, Woods Hole, USA.
Huston, A.A. (1994). *Biological diversity: The coexistence of species on changing landscape*. Cambridge University Press Cambridge.
Huston, A.A. (1997). Hidden treatments in ecological experiments: re-evaluating the ecosystem function of biodiversity. *Oecologia* 110, 449-460.
Huston, M.A., Aarssen L.W., Austin M.P., Cade B.S., Fridley J.D., Garnier E., Grime J.P., Hodgson J., Lauenroth W.K., Thompson K., Vandermeer J.H. and Wardle D.A. (2000). No consistent effect of plant diversity on productivity. *Science* 289, 1255a.
Imhoff, M.L., Bounoua L., Ricketts T., Loucks C., Harriss R. and Lawrence W.T. (2004). Global patterns in human consumption of net primary productivity. *Nature* 429, 870-873.

Intergovernmental Panel on Climate Change (2000a). *SRES Marker Scenarios*. IPCC Special Report on Emission Scenarios, Data Version 1.1, Internet site http://sres.ciesin.org/, Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Intergovernmental Panel on Climate Change (2000b). *Emission scenarios: A special report of Working Group III*. Intergovernmental Panel on Climate Change, Geneva, Switzerland.

Intergovernmental Panel on Climate Change (2007). *IPCC Fourth Assessment Report*. Internet site http://www.ipcc.ch/, Intergovernmental Panel on Climate Change, Geneva, Switzerland.

International Energy Agency (2004). *CO_2 emissions from fuel combustion 1960-2000 - Reference approach*. Internet site http://www.iea.org/w/bookshop/add.aspx?id=36, OECD/IEA, Paris, France.

International Soil Reference and Information Centre (ISRIC) and United Nations Environment Programme (2000). *Global Assessment of Soil Degradation (GLASOD)*.

IPCC Working Group II (2007). *Climate Change Impacts, Adaptation and Vulnerability. Summary for Policymakers*. Contribution to the Intergovernmental Panel on Climate Change Fourth Assessment Report - Climate Change 2007, Intergovernmental Panel on Climate Change, Brussels, Belgium.

Jarass, L., Nießlein E. and Obermair G.M. (1989). *Von der Sozialkostentheorie zum umweltpolitischen Steuerungsinstrument - Boden- und Raumbelastung durch Hochspannungsleitungen*. Nomos Baden-Baden, Germany.

Jolliet, O. and Müller-Wenk R. (2004). The LCIA Midpoint-damage Framework of the UNEP/SETAC Life Cycle Initiative. *International Journal of Life Cycle Assessment* 9(6), 394-404.

Keeling, C.D. and Whorf T.P. (2005). *Atmospheric CO2 records from sites in the SIO air sampling netwok - weekly averages derived from continuous air samples for the Mauna Loa Observatory, Hawaii, U.S.A.* Internet site http://cdiac.ornl.gov/trends/co2/sio-mlo.htm, Carbon Dioxide Research Group, Scripps Institution of Oceanography, University of California, San Diego, La Jolla, USA.

Kerr, J.T. and Currie D.J. (1995). Effects of human activity on global extinction risk. *Conservation Biology* 9(5), 1528-1538.

Klüppel, H.-J. (1998). ISO 14041. *International Journal of Life Cycle Assessment* 3(6), 301.

Knoepfel, I. (1995). *Indikatorensystem für die ökologische Bewertung des Transports von Energie*. Dissertation Nr. 11146, ETH, Zürich, Switzerland.

Köllner, T. (2000). Species-pool effect potentials (SPEP) as a yardstick to evaluate land-use impacts on biodiversity. *Journal of Cleaner Production* 8(4), 293-311.

Kreft, H. and Jetz W. (2007). Global patterns and determinants of vascular plant diversity. *Proceedings of the National Academy of Sciences of the United States of America* 104, 5925-5930.

Krewitt, W. (2002). External cost of energy - do the answers match the questions? Looking back at 10 years of ExternE. *Energy Policy* 30, 839-848.

Kristofersson, D. and Anderson J.L. (2006). Is there a relationship between fisheries and farming? Interdependence of fisheries, animal production and aquaculture. *Marine Policy* 30, 721-725.

Kuznets, S. (1955). Economic growth and income inequality. *American Economic Review* 1, 1-28.

Lassey, K.R. (2007). Livestock methane emission: From the individual grazing animal through national inventories to the global methane cycle. *Agricultural and Forest Meteorology* 142, 120-132.

Lawton, J.H. (1994). What do species do in ecosystems? *Oikos* 71, 367-374.

Lecouls, H. (1999). ISO 14043. *International Journal of Life Cycle Assessment* 4(5), 245.

Lenzen, M. (1998). The energy and greenhouse gas cost of living for Australia during 1993-94. *Energy* 23(6), 497-516.

Lenzen, M. (2001). Errors in conventional and input-output-based life-cycle inventories. *Journal of Industrial Ecology* 4(4), 127-148.

Lenzen, M. (2002). A guide for compiling inventories in hybrid LCA: some Australian results. *Journal of Cleaner Production* 10, 545-572.

Lenzen, M. (2005). Uncertainty of end-point impact measures: implications for decision-making. *International Journal of Life-Cycle Assessment*, in press.

Lenzen, M. (2006a). Structural Decomposition Analysis and the Mean-Rate-of-Change Index. *Applied Energy* 83, 185–198.

Lenzen, M. (2006b). Structural Path Analysis of ecosystem networks. *Ecological Modelling* 200, 334-342.

Lenzen, M., Borgstrom Hansson C. and Bond S. (2007a). On the bioproductivity and land disturbance metrics of the Ecological Footprint. *Ecological Economics* 61(1), 6-10.

Lenzen, M., Dey C. and Foran B. (2004a). Energy requirements of Sydney households. *Ecological Economics* 49(3), 375-399.

Lenzen, M., Dey C.J. and Murray S.A. (2004b). Historical accountability and cumulative impacts: the treatment of time in corporate sustainability reporting. *Ecological Economics* 51, 237-250.

Lenzen, M., Lane A., Widmer-Cooper A. and Williams M. (2007b). Effects of land use on threatened species. *TRENDS in Ecology and Evolution*, submitted.

Lenzen, M., Lundie S., Bransgrove G., Charet L. and Sack F. (2003). Assessing the ecological footprint of a large metropolitan water supplier - lessons for water management and planning towards sustainability. *Journal of Environmental Planning and Management* 46(1), 113-141.
Lenzen, M. and Murray S.A. (2001). A modified ecological footprint method and its application to Australia. *Ecological Economics* 37(2), 229-255.
Lenzen, M. and Murray S.A. (2003). *The ecological footprint - issues and trends*. ISA Research Paper 01-03, Internet site http://www.isa.org.usyd.edu.au/publications/documents/Ecological_Footprint_Issues_and_Trends.pdf, ISA The University of Sydney, Sydney, Australia.
Lenzen, M. and Treloar G. (2002). Energy embodied in buildings: wood versus concrete. *Energy Policy* 30(3), 249-255.
Lenzen, M., Wier M., Cohen C., Hayami H., Pachauri S. and Schaeffer R. (2006). A comparative multivariate analysis of household energy requirements in Australia, Brazil, Denmark, India and Japan. *Energy* 31, 181-207.
Leontief, W. (1986). *Input-Output Economics*. Oxford University Press New York, NY, USA.
Levett, R. (1998). Footprinting: a great step forward, but tread carefully. *Local Environment* 3(1), 67-74.
Levine, J.M., Vila M., D'Antonio C.M., Dukes J.S., Grigulis K. and Lavorel S. (2003). Mechanisms underlying the impacts of exotic plant invasions. *Proceedings of the Royal Society of London. Series B* 270, 775-781.
Lindeijer, E. (2000a). Review of land use impact methodologies. *Journal of Cleaner Production* 8, 273-281.
Lindeijer, E. (2000b). Biodiversity and life support impacts of land use in LCA. *Journal of Cleaner Production* 8, 313-319.
Livestock Information and Policy Branch (2007). *Global Livestock Production and Health Atlas*. Internet site http://www.fao.org/ag/aga/glipha, FAO Animal Production and Health Division, Rome, Italy.
Loh, J. (2006). *Living Planet Report*. Internet site http://www.panda.org/news_facts/publications/living_planet_report/lp_2006/index.cfm, WWF - World Wide Fund for Nature, Gland, Switzerland.
Long, S.P., Ainsworth E.A., Leakey A.D., Nosberger J. and Ort D.R. (2006). Food for thought: Lower-than-expected crop yield stimulation with rising CO_2 concentrations. *Science* 312, 1918-1921.
Long, S.P., Ainsworth E.A., Leakey A.D.B., Ort D.R., Nosberger J. and Schimel D. (2007). Crop models, CO_2, and climate change - Response. *Science* 315, 459-460.
Loreau, M. (1998). Biodiversity and ecosystem functioning: a mechanistic model. *Proceedings of the National Academy of Sciences of the USA* 95, 5632-5636.
Loreau, M., Naeem S., Inchausti P., Bengtsson J., Grime J.P., Hector A., Hooper D.U., Huston M.A., Raffaelli D., Schmid B., Tilman D. and Wardle D.A. (2001). Biodiversity and ecosystem functioning: Current knowledge and future challenges. *Science* 294, 804-808.
Luck, M.A., Jenerette G.D., Wu J. and Grimm N.B. (2001). The urban funnel model and the spatially heterogeneous ecological footprint. *Ecosystems* 4, 782-796.
MA (2005). *Millennium Ecosystem Assessment. Ecosystems and human well-being. Current state and trends.* Island Press Washington, D.C.
MacGillivray, C.W. and Grime J.P. (1995). Testing predictions of the resistance and resilience of vegetation subjected to extreme events. *Functional Ecology* 9, 640-649.
Malcolm, J.R., Liu C.R., Neilson R.P., Hansen L. and Hannah L. (2006). Global warming and extinctions of endemic species from biodiversity hotspots. *Conservation Biology* 20, 538-548.
Mann, C.C. (1999). Future Food: crop scientists seek a new revolution. *Science* 283, 310-314.
Marland, G., Boden T.A. and Andres R.J. (2006). *Global, Regional, and National Fossil Fuel CO2 Emissions*. Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy, Oak Ridge, U.S.A.
Marsmann, M. (2000). The ISO 14040 family. *International Journal of Life Cycle Assessment* 5(6), 317-318.
Mattison, E.H.A. and Norris K. (2005). Bridging the gaps between agricultural policy, land-use and biodiversity. *TRENDS in Ecology and Evolution* 20(11), 610-616.
Mattsson, B., Cederberg C. and Blix L. (2000). Agricultural land use in life cycle assessment (LCA): case studies of three vegetable oil crops. *Journal of Cleaner Production* 8, 283-292.
McDonald, G. and Patterson M. (2003). *Ecological Footprints of New Zealand and its Regions*. Internet site http://www.mfe.govt.nz/publications/ser/eco-footprint-sep03/index.html, Ministry for Environment New Zealand, Wellington, New Zealand.
McGrady-Steed, J., Harris P.M. and Morin P.J. (1997). Biodiversity regulates ecosystem predictability. *Nature* 390, 162-165.
McNaughton, S.J. (1977). Diversity and stability of ecological communitiesL a comment on the role of empiricism in ecology. *American Naturalist* 111, 5151-5525.

McNaughton, S.J. (1993). Biodiversity and function of grazing ecosystems. In: Schulze, E.D. and Mooney, H.A., Eds. *Biodiversity and ecosystem function*. Springer-Verlag, Berlin, Germany, 361-383.
McPherson, M.A. and Nieswiadomy M.L. (2005). Environmental Kuznets curve: threatened species and spatial effects. *Ecological Economics* 55, 395-407.
Meira, L.G. and Miguez J.D. (2000). *Note on the time-dependent relationship between emissions of greenhouse gases and climate change*. Technical Note, Internet site http://www.mct.gov.br/clima/ingles/quioto/propbra.htm, Ministry of Science and Technology, Brasília, Brazil.
Menzel, A., Sparks T.H., Estrella N., Koch E., Aasa A., Aha R., Alm-Kubler K., Bissolli P., Braslavska O., Briede A., Chmielewski F.M., Crepinsek Z., Curnel Y., Dahl A., Defila C., Donnelly A., Filella Y., Jatcza K., Mage F., Mestre A., Nordli O., Penuelas J., Pirinen P., Remisova V., Scheifinger H., Striz M., Susnik A., Van Vliet A.J.H., Wielgolaski F.E., Zach S. and Zust A. (2006). European phenological response to climate change matches the warming pattern. *Global Change Biology* 12, 1969-1976.
Mielnik, O. and Goldemberg J. (1999). The evolution of the "carbonization index" in developing countries. *Energy Policy* 27(5), 307-308.
Milà i Canals, L., Bauer C., Depestele J., Dubreuil A., Freiermuth-Knuchel R., Gaillard G., Michelsen O., Müller-Wenk R. and Rydgren B. (2007). Key elements in a framework for land use impact assessment in LCA. *International Journal of Life-Cycle Assessment* 12(1), 5-15.
Morales, P., Hickler T., Rowell D.P., Smith B. and Sykes M. (2007). Changes in European ecosystem productivity and carbon balance driven by regional climate model output. *Global Change Biology* 13, 108-122.
Mulder, C.P.H., Uliassi D.D. and Doak D.F. (2001). Physical stress and diversity-productivity relationships: the role of positive interactions. *Proceedings of the National Academy of Sciences of the United States of America* 98, 6704-6708.
Müller-Wenk, R. (1998). *Land use - the main threat to species*. IWÖ Diskussionsbeitrag Nr. 64, Institut für Wirtschaft und Ökologie, Universität St.Gallen, St.Gallen, Switzerland.
Muradian, R., O'Connor M. and Martinez-Alier J. (2002). Embodied pollution in trade: estimating the 'environmental load displacement' of industrialised countries. *Ecological Economics* 41, 51-67.
Naeem, S. (1998). Species redundancy and ecosystem reliability. *Conservation Biology* 12, 39-45.
Naeem, S. (2000). Reply to Wardle et al. *Bulletin of the Ecological Society of America* July, 241-246.
Naeem, S. (2002). Ecosystem consequences of biodiversity loss: the evolution of a paradigm. *Ecology* 83, 1537-1552.
Naeem, S., Chapin F.S., Costanza R., Ehrlich P.R., Golley F.B., Hooper D.U., Lawton J.H., O'Neill R.V., Mooney H.A., Sala O.E., Symstad A.J. and Tilman D. (1999). Biodiversity and ecosystem functioning: Maintaining natural life support processes. *Issues in Ecology* 4, 1-11.
Naeem, S. and Li S. (1997). Biodiversity enhances ecosystem reliability. *Nature* 390, 507-509.
Naeem, S., Thompson L.J., Lawler S.P. and Lawton J.H. (1995). Empirical evidence that declinig species diversity may alter the performance of terrestrial ecosystems. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences* 347, 249-262.
Naeem, S., Thompson L.J., Lawler S.P., Lawton J.H. and Woodfin R.M. (1994). Declining biodiversity can alter the performance of ecosystems. *Nature* 368, 734-737.
Naidoo, R. and Adamowicz W.L. (2001). Effects of economic prosperity on numbers of threatened species. *Conservation Biology* 15(4), 1021-1029.
Ohlemüller, R., Gritti E.S., Sykes M.T. and Thomas C.D. (2006). Towards European climate risk surfaces: the extent and distribution of analogous and non-analogous climates 1931-2100. *Global Ecology and Biogeography* 15, 395-405.
Opschoor, H. (2000). The ecological footprint: measuring rod or metaphor? *Ecological Economics* 32, 363-365.
Pandit, R. and Laband D.N. (2007a). Spatial autocorrelation in country-level models of species imperilment. *Ecological Economics* 60, 526-532.
Pandit, R. and Laband D.N. (2007b). General and specific spatial autocorrelation: insights from country-level analysis of species imperilment. *Ecological Economics* 61, 75-80.
Pearson, P.J.G. (1994). Energy, externalities and environmental quality: will development cure the ills it creates? *Energy Studies Review* 6(3), 199-216.
Pearson, R.G. and Dawson T.P. (2003). Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? *Global Ecology and Biogeography* 12, 361-371.
Peet, N.J., Carter A.J. and Baines J.T. (1985). Energy in the New Zealand household, 1974-1980. *Energy* 10(11), 1197-1208.
Pengue, W.A. (2004). Transgenic crops in Argentina and its hidden costs. In: Ortega, E. and Ulgiati, S., Eds. *Proceedings of IV Biennial International Workshop "Advances in Energy Studies"*. Unicamp, Campinas, SP, Brazil, 91-101.

Peters, G., Sack F., Lenzen M., Lundie S. and Gallego B. (2007). A novel Ecological Footprint calculation for the Australian water industry: Regionalisation and inclusion of downstream impacts. *Journal of Applied Input-Output Analysis*, in press.

Pimentel, D., Terhune E.C., Dyson-Hudson R., Rochereau S., Samis R., Smith E.A., Denman D., Reifschneider D. and Shepard M. (1976). Land degradation: effects on food and energy resources. *Science* 194(261), 149-155.

Portney, P.R. (1994). The Contingent Valuation debate: why economists should care. *Journal of Economic Perspectives* 8(4), 3-17.

Poschlod, P., Bakker J.P. and Kahmen S. (2005). Changing land use and its impact on biodiversity. *Basic and Applied Ecology* 6, 93-98.

Power, M.E., Tilman D., Estes J.A., Menge B.A., Bond W.J., Mills L.S., Daily G., Castilla J.C., Lubchenco J. and Paine R.T. (1996). Challenges in the quest for keystones. *BioScience* 46, 609-620.

Prentice, I.C., Cramer W., Harrison S.P., Leemans R., Monserud R.A. and Solomon A.M. (1992). A global biome model based on plant physiology and dominance, soil properties and climate. *Journal of Biogeography* 19, 117-134.

Purvis, A. and Hector A. (2000). Getting the measure of biodiversity. *Nature* 405, 212-219.

Rabl, A. and Spadaro J.V. (1999). Damages and costs of air pollution: an analysis of uncertainties. *Environment International* 25(1), 29-46.

Rabl, A. and Spadaro J.V., Eds. (2005). *Extension of accounting framework and Policy Applications*. ExternE - Externalities of Energy. Directorate-General for Research, Sustainable Energy Systems, Luxembourg-Ville, Luxembourg.

Rahbek, C., Gotelli N.J., Colwell R.K., Entsminger G.L., Rangel T. and Graves G.R. (2007). Predicting continental-scale patterns of bird species richness with spatially explicit models. *Proceedings Of The Royal Society B-Biological Sciences* 274, 165-174.

Rapport, D.J. (2000). Ecological footprints and ecosystem health: complementary approaches to a sustainable future. *Ecological Economics* 32, 381-383.

Rees, W.E. (1992). Ecological footprints and appropriated carrying capacity: what urban economics leaves out. *Environment and Urbanization* 4(2), 121-130.

Reidsma, P., Tekelenburg T., van den Berg M. and Alkemade R. (2006). Impacts of land-use changes on biodiversity: An assessment of agricultural biodiversity in the European Union. *Agriculture, Ecosystems and Environment* 114, 86-102.

Reynolds, A.M. and O'Doherty J.V. (2006). The effect of amino acid restriction during the grower phase on compensatory growth, carcass composition and nitrogen utilisationin grower-finisher pigs. *Livestock Science* 104, 112-120.

Robert, L., Hirsch R.L., Bezdek R. and Wendling R. (2006). Peaking of world oil production and its mitigation. *American Institute of Chemical Engineers Journal* 52, 2-8.

Roberts, P.C. (1975). Energy analysis in modelling. In: Blair, I.M., Jones, B.D. and Van Horn, A.J., Eds. *Aspects of energy conversion*. Pergamon Press, Oxford, UK, 759-771.

Rosa, L.P. and Ribeiro S.K. (2001). The present, past, and future contributions to global warming of CO_2 emissions from fuels. *Climatic Change* 48(2), 289-307.

Rosa, L.P., Ribeiro S.K., Muylaert M.S. and de Campos C.P. (2004). Comments on the Brazilian Proposal and contributions to global temperature increase with different climate responses - CO_2 emissions due to fossil fuels, CO_2 emissions due to land use changes. *Energy Policy* 32(13), 1499-1510.

Rosa, L.P. and Schaeffer R. (1993). Greenhouse gas emissions from hydroelectric reservoirs. *Ambio* 22(4), 164-165.

Rosa, L.P. and Schaeffer R. (1995). Global warming potentials - the case of emissions from dams. *Energy Policy* 23(2), 149-158.

Rosegrant, M.W. and Ringler C. (1997). World food markets into the 21st century: Environmental and resource constraints and policies. *Australian Journal of Agricultural and Resource Economics* 41(3), 401-428.

Rounsevell, M.D.A., Ewert F., Reginster I., Leemans R. and Carter T.R. (2005). Future scenarios of European agricultural land use - II. Projecting changes in cropland and grassland. *Agriculture, Ecosystems and Environment* 107, 117-135.

Rounsevell, M.D.A., Reginster I., Araújo M.B., Carter T.R., Dendoncker N., Ewert F., House J.I., Kankaanpää S., Leemans R., Metzger M.J., Schmit C., Smith P. and Tuck G. (2006). A coherent set of future land use change scenarios for Europe. *Agriculture, Ecosystems and Environment* 114, 57-68.

Ryding, S.-O. (1999). ISO 14042. *International Journal of Life Cycle Assessment* 4(6), 307.

Sala, O.E., Chapin F.S., Armesto J.J., Berlow E., Bloomfield J., Dirzo R., Huber-Sanwald E., Huenneke L.F., Jackson R.B., Kinzig A., Leemans R., Lodge D.M., Mooney H.A., Oesterheld M., Poff N.L., Sykes M.T., Walker B.H., Walker M. and Wall M. (2000). Global biodiversity scenarios for the year 2100. *Science* 287, 1770-1774.

Schenck, R.C. (2001). Land use and biodiversity indicators for Life Cycle Impact Assessment. *International Journal of Life Cycle Assessment* 6(2), 114-117.
Selden, T.M. and Song D. (1994). Environmental quality and development: is there a Kuznets curve for air pollution emissions? *Journal of Environmental Economics and Management* 27, 147-162.
Shafik, N. (1994). Economic development and environmental quality: an econometric analysis. *Oxford Economic Papers* 46, 757-773.
Simmons, C., Lewis K. and Barrett J. (2000). Two feet - two approaches: a component-based model of ecological footprinting. *Ecological Economics* 32, 375-380.
Simpson, E.H. (1949). Measurement of diversity. *Nature* 163, 688.
Singer, P. and Mason J. (2006). *The Way We Eat: Why Our Food Choices Matter*. Rodale Press New York, USA.
Smil, V. (2004). *Enriching the Earth: Fritz Haber, Carl Bosch and the Transformation of World Food Production*. MIT Press Cambridge, USA.
Spangenberg, J.H. (2007). Biodiversity pressure and the driving forces behind. *Ecological Economics* 61, 146-158.
Steele, L.P., Krummel P.B. and Langenfelds R.L. (2003). *Atmospheric CH_4 concentrations (ppbv) derived from flask samples collected at Macquarie Island, Australia*. Internet site http://cdiac.ornl.gov/trends/atm_meth/csiro/csiro-macqch4.html, Commonwealth Scientific and Industrial Research Organisation (CSIRO), Aspendale, Australia.
Stern, D.I. (2003). *The rise and fall of the Environmental Kuznets Curve*. Working Papers 0302, Rensselaer Polytechnic Institute, Troy, NY, USA.
Stern, D.I., Common M.S. and Barbier E.B. (1996). Economic growth and environmental degradation: the Environmental Kuznets Curve and sustainable development. *World Development* 24(7), 1151-1160.
Stern, D.I. and Kaufmann R.K. (1998). *Global historical anthropogenic CH_4 emissions expressed in Tg*. Internet site http://cdiac.ornl.gov/trends/meth/ch4.htm, Center for Energy and Environmental Studies, Boston University, Boston, USA.
Stöglehner, G. (2003). Ecological footprint - a tool for assessing sustainable energy supplies. *Journal of Cleaner Production* 11, 267-277.
Sun, J.W. and Ang B.W. (2000). Some properties of an exact energy decomposition model. *Energy* 25, 1177-1188.
Swan, G. (1998). *Evaluation of land use in Life Cycle Assessment*. CPM Report 1998:2, Center for Environmental Assessment of Product and Material Systems, Chalmers University of Technology, Göteborg, Sweden.
Swan, G. and Pettersson B. (1998). Land use evaluation in forestry. In: Swan, G., Ed. *Evaluation of land use in Life Cycle Assessment, Center for Environmental Assessment of Product and Material Systems, CPM Report 1998:2*. Chalmers University of Technology, Göteborg, Sweden, 16-21.
Task Force on National Greenhouse Gas Inventories (1996). *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories - Reporting Instructions (Volume 1)*. http://www.ipcc-nggip.iges.or.jp/public/gl/invs1.htm, Institute for Global Environmental Strategies, Intergovernmental Panel on Climate Change - IPCC National Greenhouse Gas Inventories Programme, Tokyo, Japan.
The IUCN Red List of Threatened Species (2006). *Summary statistics for globally threatened species. Table 5: Threatened species in each country (totals by taxonomic group)*. Internet site http://www.iucnredlist.org/info/stats, International Union for the Conservation of Nature and Natural Resources (IUCN), Cambridge, UK.
Thomas, C.D., Cameron A., Green R.E., Bakkenes M., Beaumont L.J., Collingham Y.C., Erasmus B.F.N., de Siqueira M.F., Grainger A., Hannah L., Hughes L., Huntley B., van Jaarsveld A.S., Midgley G.F., Miles L., Ortega-Huerta M.A., Peterson A.T., Phillips O.L. and Williams S.E. (2004). Extinction risk from climate change. *Nature* 427, 145-148.
Thuiller, W., Lavorel S., Araújo M.B., Sykes M.T. and Prentice I.C. (2005). Climate change threats to plant diversity in Europe. *Proceedings Of The National Academy Of Sciences Of The United States Of America* 102, 8245-8250.
Thuiller, W., Midgley G.F., Hughes G.O., Bomhard B., Drew G., Rutherford M.C. and Woodward F.I. (2006). Endemic species and ecosystem sensitivity to climate change in Namibia. *Global Change Biology* 12, 759-776.
Tilman, D. (1999). Diversity and production in European grasslands. *Science* 286, 1099-1100.
Tilman, D. (2000). What *Issues in Ecology* is, and isn't. *Bulletin of the Ecological Society of America* July, 240.
Tilman, D., Cassman K.G., Matson P.A., Naylor R. and Polasky S. (2002). Agricultural sustainability and intensive production practices. *Nature* 418, 671-677.
Tilman, D. and Downing J.A. (1994). Biodiversity and stablility in grasslands. *Nature* 367, 363-365.

Tilman, D., Fargione J., Wolff B., D'Antonio C., Dobson A., Howarth R., Schindler D., Schlesinger W.H., Simberloff D. and Swackhamer D. (2001a). Forecasting agriculturally driven global environmental change. *Science* 292, 281-284.
Tilman, D., Knops J., Wedin D., Reich P., Ritchie M. and Siemann E.S. (1997a). The influence of functional diversity and composition on ecosystem processes. *Science* 277, 1300-1302.
Tilman, D., Lehman C.L. and Thomson K.T. (1997b). Plant diversity and ecosystem productivity: Theoretical considerations. *Proceedings of the National Academy of Sciences of the United States of America* 94, 1857-1861.
Tilman, D., May R.M., Lehman C.L. and Nowak M.A. (1994). Habitat destruction and the extinction debt. *Nature* 371, 65-66h.
Tilman, D., Reich P.B., Knops J., Wedin D., Mielke T. and Lehman C. (2001b). Diversity and productivity in a long-term grassland experiment. *Science* 294, 843-845.
Tilman, D., Wedin D. and Knops J. (1996). Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* 379, 718-720.
Treloar, G. (1997). Extracting embodied energy paths from input-output tables: towards an input-output-based hybrid energy analysis method. *Economic Systems Research* 9(4), 375-391.
Turner, K., Wiedmann T., Lenzen M. and Barrett J. (2007). Examining the global environmental impact of regional consumption activities—Part 1: A technical note on combining input–output and ecological footprint analysis. *Ecological Economics* 62(1), 37-44.
U.S. Census Bureau (2006). *Table 001: Total Midyear Population*. Internet site http://www.census.gov/ipc/www/idbsprd.html, U.S. Department of Commerce, Washington DC.
Udo de Haes, H.A. (2006). How to approach land use in LCIA or, how to avoid the Cinderella effect? *International Journal of Life Cycle Assessment* 11(4), 219-221.
Udo de Haes, H.A. and Heijungs R. (2004). Three strategies to overcome the limitations of Life-Cycle Assessment. *Journal of Industrial Ecology* 8(3), 19-32.
Udo de Haes, H.A., Jolliet O., Finnveden G., Hauschild M., Krewitt W. and Mueller-Wenk R. (1999). Best available practice regarding impact categories and category indicators in life cycle impact assessment. *International Journal of Life Cycle Assessment* 4(2), 1-15.
United Nations Statistics Division (2007a). *UN comtrade - United Nations Commodity Trade Statistics Database*. Internet site http://comtrade.un.org/, United Nations Statistics Division, UNSD, New York, USA.
United Nations Statistics Division (2007b). *National Accounts Main Aggregates Database*. Internet site http://unstats.un.org/unsd/snaama/Introduction.asp, United Nations Statistics Division (UNSD), New York, USA.
van den Bergh, J.C.J.M. and Verbruggen H. (1999). Spatial sustainability, trade and indicators: an evaluation of the 'ecological footprint'. *Ecological Economics* 29(1), 61-72.
van Dobben, H.F., Schouwenberg E.P.A.G., Nabuurs G.J. and Prins A.H. (1998). Biodiversity and productivity parameters as a basis for evaluating land use changes in LCA. In: IVAM Environmental Research, Ed. *Biodiversity and life support indicators for land use impacts, Publication Series Raw Materials Nr 1998/07*. Ministry of Transport, Public Works and Watermanagement, Delft, Netherlands, Annex 1.1-1.50.
van Kooten, G.C. and Bulte E.H. (2000). The ecological footprint: useful science or politics? *Ecological Economics* 32, 385-389.
van Vuuren, D.P. and Bouwman L.F. (2005). Exploring past and future changes in the ecological footprint for world regions. *Ecological Economics* 52(1), 43-62.
van Vuuren, D.P., Sala O.E. and Pereira H.M. (2006). The future of vascular plant diversity under four global scenarios. *Ecology And Society* 11, 15.
Venetoulis, J. and Talberth J. (2006). *Net Primary Productivity as a basis for Ecological Footprint analysis*. Redefining Progress, Oakland, USA.
Venetoulis, J. and Talberth J. (2007). Refining the Ecological Footprint. *Environment, Development and Sustainability*, in press.
Vitousek, P.M. and Hooper D.U. (1993). Biological diversity and terrestrial ecosystem biogeochemistry. In: Schulze, E.-D. and Mooney, H.A., Eds. *Biodiversity and ecosystem function*. Springer-Verlag, Berlin, Germany, 3-14.
Vitousek, P.M., Mooney H.A., Lubchenco J. and Melillo J.M. (1997). Human dominantion of Earth's ecosystems. *Science* 277, 494-499.
Voigtländer, J.G., Lindeijer E., Witte J.-P.M. and Hendriks C. (2004). Characterising the change of land-use based on flora: application for EIA and LCA. *Journal of Cleaner Production* 12, 47-57.
Vringer, K. and Blok K. (1995). The direct and indirect energy requirements of households in the Netherlands. *Energy Policy* 23(10), 893-910.

Wackernagel, M. (1994). *Ecological Footprint and Appropriated Carrying Capacity: A Tool for Planning Toward Sustainability*. PhD Thesis, The University of British Columbia, Vancouver, Canada.

Wackernagel, M. (2000). Importing carrying capacity: how global trade enables nations and the world to accumulate an ecological debt. In: Watt, K., Ed. *Encyclopedia on Human Ecology*. Transaction Publishers, Rutgers, NJ, USA.

Wackernagel, M., Lewan L. and Hansson C.B. (1999). Evaluating the use of natural capital with the ecological footprint. *Ambio* 28(7), 604-612.

Wackernagel, M., Monfreda C., Moran D., Wermer P., Goldfinger S., Deumling D. and Murray M. (2005). *National Footprint and Biocapacity Accounts 2005: The underlying calculation method*. Global Footprint Network, Oakland, USA.

Wackernagel, M., Schulz N.B., Deumling D., Callejas Linares A., Jenkins M., Kapos V., Monfreda C., Loh J., Myers N., Norgaard R. and Randers J. (2002). Tracking the ecological overshoot of the human economy. *Proceedings of the National Academy of Sciences* 99(14), 9266-9271.

Wagendorp, T., Gulinck H., Coppin P. and Muys B. (2006). Land use impact evaluation in life cycle assessment based on thermodynamics. *Energy* 31, 112-125.

Wardle, D.A. (1999). Is "Sampling Effect" a problem for experiments investigating biodiversity-ecosystem function relationships? *Oikos* 87, 403-407.

Wardle, D.A., Huston M.A., Grime J.P., Berendse F., Garnier E., Lauenroth W.K., Setälä H. and Wilson S.G. (2000). Biodiversity and ecosystem function: an issue in ecology. *Bulletin of the Ecological Society of America* July, 235-239.

Weidema, B. and Lindeijer E. (2001). *Physical impacts of land use in product life cycle assessment*. Final Report of the EURENVIRON-LCAGAPS sub-project on land use, Department of Manufacturing, Engineering and Management, Technical University of Denmark, Lyngby, Denmark.

Whittaker, R.H. and Likens G.E. (1973). Primary production: the biosphere and man. *Human Ecology* I(4), 357.

Wiedmann, T. and Lenzen M. (2007). On the conversion from local hectares to global hectares in Ecological Footprint analysis. *Ecological Economics* 60(4), 673-677.

Wier, M., Lenzen M., Munksgaard J. and Smed S. (2001). Environmental effects of household consumption pattern and lifestyle. *Economic Systems Research* 13(3), 259-274.

Williams, J.W., Jackson S.T. and Kutzbach J.E. (2007). Projected distributions of novel and disappearing climates by 2100 AD. *Proceedings of the National Academy of Sciences of the USA* 104(14), 5738–5742.

Witte, J.P.M. and van der Meijden R. (1995). Verspreidingskaarten van de botanische kwaliteit in Nederland uit FLORBASE. *Gorteria 21* 1/2, 3-60.

Wollenweber, B., Porter J.R. and Lubberstedt T. (2005). Need for multidisciplinary research towards a second green revolution. *Current Opinion in Plant Biology* 8, 337-341.

World Organisation for Animal Health (2007). *Update on avian influenza in animals (Type H5)*. Internet site http://www.oie.int/downld/AVIAN%20INFLUENZA/A_AI-Asia.htm, World Organisation for Animal Health, Paris, France.

World Resources Institute (2005a). *WRI Biodiversity Overview*. Internet site http://earthtrends.wri.org/pdf_library/data_tables/bio1_2005.pdf, World Resources Institute

World Resources Institute (2005b). *Land area classification by ecosystem type*. Internet site http://earthtrends.wri.org/pdf_library/data_tables/for2_2003.pdf, World Resources Institute

World Resources Institute (2006). Climate Analysis Indicators Tool (CAIT) on-line database version 3.0. Internet site http://cait.wri.org, World Resources Institute, Washington D.C., USA.

Appendix: List of countries

Afghanistan
Albania
Algeria
American Samoa
Andorra
Angola
Anguilla
Antigua and Barbuda
Argentina
Armenia
Aruba
Australia
Austria
Azerbaijan
Bahamas
Bahrain
Bangladesh
Barbados
Belarus
Belgium
Belize
Benin
Bermuda
Bhutan
Bolivia
Bosnia and Herzegovina
Botswana
Bouvet Island
Brit Ind Ocean Terr
Virgin Islands (Brit)
Brazil
Brunei Darussalam
Bulgaria
Burkina Faso
Burundi
Côte d'Ivoire
Cambodia
Cameroon
Canada
Cape Verde
Cayman Islands
Central African Rep
Chad
Chile
China
Hong Kong
Macau
Colombia
Comoros
Congo, Republic of
Cook Islands
Costa Rica
Croatia
Cuba
Cyprus
Czech Republic
Korea, Dem People's Rep
Congo, Dem Rep
Denmark
Djibouti
Dominica
Dominican Republic
Ecuador
Egypt
El Salvador
Equatorial Guinea
Eritrea
Estonia
Ethiopia
Faroe Islands
Falkland Is Malvinas
Fiji
Finland
France
French Polynesia
Micronesia, Fed States
Gabon
Gambia
Georgia
Germany
Ghana
Gibraltar
Greece
Greenland
Grenada
Guam
Guatemala
Guinea
Guinea-Bissau
Guyana
Haiti
Honduras
Hungary
Iceland
India
Indonesia
Iran
Iraq
Ireland
Israel
Italy
Jamaica
Japan
Jordan
Kazakhstan
Kenya
Kiribati
Kuwait
Kyrgyzstan
Laos
Latvia
Lebanon
Lesotho
Liberia
Libya
Lithuania
Luxembourg
Madagascar
Malawi
Malaysia
Maldives
Mali
Malta
Marshall Islands
Mauritania
Mauritius
Mexico
Mongolia
Montserrat
Morocco
Mozambique
Myanmar
Northern Mariana Is
Namibia
Nauru
Nepal
Netherlands Antilles
Netherlands
New Caledonia
New Zealand
Nicaragua
Niger
Nigeria
Niue
Norfolk Island
Norway
Palestine, Occupied
Oman
Pakistan
Palau
Panama
Papua New Guinea
Paraguay
Peru
Philippines
Pitcairn
Poland
Portugal
Qatar
Korea
Moldova
Romania
Russian Federation
Rwanda
St Helena
St Kitts and Nevis
St Lucia
St Pierre & Miquelon
St Vincent/Grenadines
Samoa
São Tomé & Príncipe
Saudi Arabia
Senegal
Serbia and Montenegro
Seychelles
Sierra Leone
Singapore
Slovakia
Slovenia
Solomon Islands
Somalia
South Africa
Spain
Sri Lanka
Sudan
Suriname
Swaziland
Sweden
Switzerland
Syria
Tajikistan
Macedonia
Thailand
Timor-Leste
Togo
Tokelau
Tonga
Trinidad and Tobago
Tunisia
Turkey
Turkmenistan
Turks and Caicos Is
Tuvalu
Uganda
Ukraine
United Arab Emirates
United Kingdom
Tanzania
Pacific Islands Trust Terr
Uruguay
United States of America
Uzbekistan
Vanuatu
Venezuela
Viet Nam
Wallis and Futuna Is
Western Sahara
Yemen
Zambia
Zimbabwe