Final report

Project code: B.CCH.2084

Prepared by: Andrew D Moore

Life Cycle Logic Pty Ltd

Date published: March 2014 (revised Feb 2015)

ISBN: 9781740362955

PUBLISHED BY
Meat & Livestock Australia Limited
Locked Bag 991
NORTH SYDNEY NSW 2059

Review of approaches to assess land use impacts on biodiversity in LCA

Meat & Livestock Australia acknowledges the matching funds provided by the Australian Government to support the research and development detailed in this publication.

Abstract

Life cycle assessment (LCA) is widely used to assess a range of environmental impacts that occur throughout the life cycle of products. Although many different environmental issues are commonly included in these studies (e.g. carbon footprint and water scarcity footprint) the impacts of land use on biodiversity have often been excluded due to reasons including a lack of widely accepted methods and data limitations. Land use is the main cause of loss of biodiversity so it is important that these impacts be included, especially in agricultural LCA studies where large areas of land are occupied. This study reviews the current methods that can be used to include biodiversity impacts in LCA and the data required to apply these methods. Two methods have been recommended to assess the biodiversity impacts of livestock supply chains in Australia. These methods were described and applied within recently published studies: “Land use impact assessment of margarine” (Canals, Rigarlsford, and Sim 2013); and “Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study” (Mueller, de Baan, and Koellner 2014). Both of the methods described in these papers follow the latest guidelines published by the UNEP-SETAC life cycle land use working group. These methods can be used to quantify the biodiversity impacts of different livestock production systems and identify hotspots which can be targeted in order to minimise biodiversity impacts.

Executive summary

The objective of this paper is to review and recommend current methods and approaches which can be used to assess land use impacts on biodiversity using life cycle assessment (LCA).

LCA is widely used to assess the environmental impacts of products using a whole-of-life perspective. Environmental impacts such as climate change, water scarcity, and resource depletion, are commonly assessed but biodiversity impacts are often excluded, despite the importance of the issue. Reasons commonly stated are a lack of widely accepted methods and data limitations. Changes in land use are the primary source of biodiversity loss worldwide (Schenck 2001; Hoekstra et al. 2004) and there is urgent need to quantify and manage these impacts using a holistic approach which LCA provides.

The common understanding of biodiversity is species richness (the number of different species in an area) but biodiversity also includes genetic diversity (diversity between individuals or populations), and ecosystem diversity (diversity between different communities and habitats). Biodiversity isn't evenly distributed across landscapes. Its spatial distribution is determined by a range of factors including climate, geology, and the evolutionary history of the planet (WWF 2014). Different types of land use (e.g. nature conservation, forest, pasture, urban area) can also have different impacts on biodiversity depending on the ecosystem type where these impacts occur (e.g. tropical forests, temperate grasslands, deserts). To accurately represent the impacts which different land uses have on biodiversity spatially differentiated characterisation factors are required. Measures of land occupation (m^2.years) have been included in the results of some LCA studies but this inventory level measure does not take into account the spatial variability of biodiversity and the impacts that land use can cause and cannot be directly compared in a meaningful way. Land use causes impacts to biodiversity through using the land (occupation), changing the land use (transformation), and the level of irreversibility of the impacts (permanence).

Over more than two decades, the issue of how to include biodiversity impacts of land use in LCA has received considerable research. The UNEP-SETAC[1] Life Cycle Initiative land use working group recently published guidelines, in a special issue of the International Journal of Life Cycle Assessment (Koellner and Geyer 2013). Almost all of the methods proposed before these UNEP-SETAC publications had limitations which prevented application outside of the regions for which they were developed or being applied to model supply chains that extended globally. The UNEP-SETAC guidelines were developed by leading experts in this field and represent the current state of consensus and description of best practice.

[1] United Nations Environment Programme (UNEP)/ Society of Environmental Toxicology and Chemistry Life Cycle Initiative (SETAC)

All current approaches to assess land use impacts on biodiversity use mapping data and analysis (using geographic information systems) to account for the uneven spatial distribution of biodiversity (as defined by biomes or ecoregions), land uses, and land cover.

Robust methods are available which can be used, for regions throughout the world, to quantify land use impacts on biodiversity for products including those which have global supply chains. The results can be used to identify hotspots in the life cycle (areas where improvements can be made) and to make meaningful comparisons between land management practices so that the impacts on biodiversity can ultimately be reduced.

This study reviewed current methods which followed UNEP-SETAC guidelines or approaches that were published since its release. It also reviewed both global and Australian spatial data that are available to implement these methods. Of the methods reviewed, two were promising and could be applied for Australian livestock supply chains. These methods were described and applied within recently published studies:

1. Land use impact assessment of margarine (Canals, Rigarlsford, and Sim 2013), as based on Land use impacts on biodiversity in LCA: a global approach (de Baan, Alkemade, and Koellner 2013); and
2. Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study (Mueller, de Baan, and Koellner 2014).

Both methods can be applied globally and can be used to identify biodiversity impact hotspots. The important difference between the two approaches is that the second approach can also be used to make comparison between farming practices, however, is a more time intensive approach to apply. The second method is preferred as it is the most comprehensive approach to assess biodiversity impacts of land use and includes measures of biodiversity on both the species and ecosystem levels (ecosystem scarcity and vulnerability).

To apply these methods to Australian livestock supply chains it is recommended to proceed using an approach which follows the LCA framework (ISO 2006): goal and scope definition, inventory analysis, impact assessment, and interpretation.

A goal and scope workshop is recommended to clearly define the goal for a pilot study. It is recommended to use a previous LCA study as a source of life cycle inventory data because this approach has been used by many biodiversity impact studies and will provide greater context for the results. Using a recent LCA study of beef supply chains (Wiedemann, Murphy, McGahan, Bonner, et al. 2013; Wiedemann, Murphy, McGahan, Renouf, et al. 2013; Ridoutt et al. 2013) is recommended.

The next step would be to conduct inventory analysis to collect information required for the life cycle and conduct a detailed search for required biological survey data (for relevant geographic regions) that would be required to apply the recommended biodiversity impact assessment methods. Impact assessment can be carried out using the impact assessment methods for which sufficient data is available. The results can then be interpreted and used to produce recommendations of how to minimise biodiversity impacts that occur throughout the supply chain.

Table of Contents

List of Figures

List of Tables

Abbreviations

ABARES	Australian Bureau of Agricultural and Resource Economics and Sciences
ABS	Australian Bureau of Statistics
ANOVA	Analysis of variance
BDP	Biodiversity damage potential
DAFF	Australian Government Department of Agriculture
EDP	Ecosystem damage potential
EIA	Environmental impact assessment
FAO	Food & Agriculture Organization of the United Nations
GIS	Geographic information systems
GLC	Global land cover
ha	Hectares
HANPP	Human appropriation of net primary production
IBRA	Interim Biogeographic Regionalisation for Australia
ISO	International Standard Organization
IUCN	International Union for Conservation of Nature
JRC	Joint Research Centre of the European Commission
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
LENZ	Land Environments of New Zealand
LULCIA	LCIA of land use working group of the UNEP-SETAC Life Cycle Initiative
m^2	One square metre
MLA	Meat and Livestock Australia
NPP	Net primary productivity
NPP_0	Net primary productivity of potential biomass (g C m^2 $year^{-1}$)
PNV	Potential natural vegetation
SETAC	Society of Environmental Toxicology and Chemistry
UNEP	United Nations Environment Programme
WDPA	World Database on Protected Areas
WWF	World Wildlife Fund

1 Introduction

Life cycle assessment (LCA) is a methodology which quantifies the potential environmental impacts over the whole life of a product, from cradle to the grave. By quantifying different environmental impacts (e.g. carbon footprint, water footprint) over the whole life cycle leads to greater understanding of the environmental performance of the products and helps to identify areas for potential improvement. LCA has been widely used for assessing the environmental impacts of agricultural products both around the world and here in Australia.

There has been a considerable investment in LCA in the Australian agricultural sector with over 75 LCA studies being undertaken between 2003 and 2013 covering some 80% of Australia’s key agricultural commodities (Renouf and Fujita-Dimas 2013). The majority of these studies include environmental issues such as global warming and energy use, with some also including water use. Few studies considered the impacts of land use (Renouf and Fujita-Dimas 2013) and even less have included impacts on biodiversity.

There is a strong need to use a life cycle approach to assess the land use impacts on biodiversity because this approach includes the whole supply chain including the impacts associated with the production of supplementary feeds (forages and grains) used for more intensive production methods. A typical Environmental Impact Assessment (EIA) approach assesses the local impacts at a particular site and does not include off-site impacts so is therefore not suitable when the whole life cycle needs to be considered (Canals et al. 2007).

Without considering all relevant environmental impacts, including biodiversity impacts, these assessments do not adhere to the LCA principle of comprehensiveness which is required to ensure that decisions do not simply shift environmental burdens between impact categories (Finkbeiner 2009; Ridoutt et al. 2013; Schenck 2001). The risk is that by making decisions to improve sustainability, without a comprehensive assessment considering all relevant environmental impacts such as land use impacts on biodiversity, it can lead to unintended outcomes.

In fact, very few of LCA studies conducted globally have included impacts of land use on biodiversity (Geyer, Stoms, et al. 2010; Penman, Law, and Ximenes 2010). Reasons for this are commonly stated as being due to the lack of widely accepted methodologies and indicators (Ridoutt et al. 2013; Mattila, Helin, and Antikainen 2011; Hauschild et al. 2013; Canals et al. 2007; Weidema and Lindeijer 2001) or lack of data required to quantify the impacts using the proposed methods for supply chains which often have global reach (Coelho and Michelsen 2013; Canals et al. 2007; de Baan, Alkemade, and Koellner 2013; Koellner, de Baan, et al. 2013). Also, few of the proposed methods are suitable for comparing different farming systems which limits the usefulness of such approaches (Weidema and Lindeijer 2001; de Baan, Alkemade, and Koellner 2013; Mattila, Helin, and

Antikainen 2011; Mueller, de Baan, and Koellner 2014). The topic also requires in depth knowledge of three distinct disciplines - LCA, geographic information systems (GIS), and ecology - which adds to the complexity of the task. It is widely recognised that there is an urgent need to include land use impacts on biodiversity in LCA (de Baan, Alkemade, and Koellner 2013; Canals et al. 2007; Koellner, de Baan, et al. 2013).

1.1 Background

1.1.1 Land use

The term “land use” simply refers to the purpose which the land has been assigned (DAFF 2006a). The land may be used for agriculture, housing, or nature conservation and each of these uses have impacts on the biodiversity qualities of the land.

The human population is currently over seven billion (United Nations 2013). Housing, feeding, and providing energy for this many people on a planet with limited land area inevitably leads to increased competition between land uses and impacts on biodiversity and the ecosystem services that the natural environment provides (biomass production, climate regulation, water purification, freshwater regulation and erosion regulation) (Koellner, de Baan, et al. 2013). This pressure has resulted in what has been termed by some authors as a biome crisis, where not only are individual species facing extinction but whole ecosystems as well (Hoekstra et al. 2004). Changes in land use are the primary source of biodiversity loss worldwide (Schenck 2001; Hoekstra et al. 2004) so there is urgent need to quantify and manage these impacts.

Land use impacts are particularly relevant for agriculture. Thirty per cent is of our planet’s land area is currently occupied for crops and pastures (FAO 2013). Globally, livestock production is the largest user of agricultural land (FAO 2009) and it is widely accepted that the livestock sector must continue to improve its environmental performance (FAO 2009; Frank Mitloehner 2012). In addition to this, a global shift towards a diet consisting of more livestock products is placing greater pressure on biodiversity thorough land use change, not only through an increase in livestock production itself, but also through link to crops used for feed production (FAO 2009).

In Australia, 56% of the continent (4.3 million square kilometres) is used for livestock grazing, although almost all of this (46% of Australia) is associated with grazing on unmodified natural vegetation (DAFF 2006a). As biodiversity distribution is spatially heterogeneous there is not a linear relationship between land use and the impacts on biodiversity. Land use impacts on biodiversity studies have demonstrated that simply occupying a larger land area does not necessarily result in higher impacts on biodiversity (Mueller, de Baan, and Koellner 2014).

1.1.2 Biodiversity

Biodiversity refers to the wide variety of living organisms in all their different forms including plants, animals, micro-organisms, fungi, and insects. So far over 1.8 million different species have been identified on earth but the total number has been estimated to be somewhere between 5 to 100 million (UNEP 2014). Biodiversity isn't evenly distributed across landscapes. Its distribution is determined by a range of factors including climate, geology, and the evolutionary history of the planet (WWF 2014). Using these factors Olson et al. (2001) classified and mapped terrestrial ecosystems globally into 14 biomes and 867 ecoregions. This biome and ecoregion mapping is commonly used in LCA approaches to quantify biodiversity impacts.

The common understanding of biodiversity is the number of different species in a particular area (known as species richness) but the concepts of biodiversity also includes genetic diversity (diversity between individuals or populations), and ecosystem diversity (diversity between different communities and habitats).

Despite the considerable efforts which have been made by the ecological scientific community to develop a single metric for biodiversity it has proven to be an elusive task (Magurran 2004). Several different methods for measuring biodiversity (for use in LCA) have therefore been proposed. These metrics generally measure biodiversity on either the species or ecosystem level. On the species level they include species richness, abundance, and evenness. Species richness is the simplest measure for biodiversity. It quantifies the number of different species identified within a particular area. Both absolute and relative measures of species richness have been used in LCA (Canals, Rigarlsford, and Sim 2013; de Baan, Alkemade, and Koellner 2013; Mueller, de Baan, and Koellner 2014). Species abundance is a measure of the absolute size of populations of a species (Geyer, Lindner, et al. 2010) and evenness is a relative measure of the level of abundance between different species present in an area (Magurran 2004). Areas may be dominated by one species (low evenness) or equally amongst species (high evenness). Approaches based on the absolute number of rare species have also been proposed (Weidema and Lindeijer 2001). There are limitations to species based approaches if they only focus on one taxonomic group (e.g. plants), although, species level data for many taxa for many areas is often incomplete (Penman, Law, and Ximenes 2010; Coelho and Michelsen 2013; Mueller, de Baan, and Koellner 2014). The species diversity in one taxonomic group (e.g. plants) can be a poor indicator for other taxonomic groups (e.g. mammals or reptiles).

Biodiversity on the ecosystem level is often represented by measures of scarcity, vulnerability, or hemeroby. The argument for using ecosystem level assessment is that in order to conserve species, the ecosystems in which they occur must also be conserved (Weidema and Lindeijer 2001).

Some ecosystem types are widely distributed (and have low scarcity) and others occupy smaller extents (and have higher scarcity). Ecosystem scarcity can be calculated as the inverse of the total potential area available for each ecosystem type. Global ecosystem scarcity data is available at the biome level. Ecosystem vulnerability is a measure of how much of an ecosystem type remains in its natural state (i.e. percentage of original habitat remaining). Ecosystems which have low percentages remaining are more vulnerable than those with higher relative amounts remaining. The relationship between number of original species which can be supported and the area remaining, however, is non- linear and is described by species-area-curves which vary depending on the ecosystem type. Hemeroby (level of human influence on a natural environment) describes the intensity of land use and the level of naturalness of an ecosystem (Brentrup et al. 2002).

1.1.3 Land use impact assessment in LCA

Over more than two decades the topic of assessing land use impacts on biodiversity using LCA has received considerable amount of research. In the earlier publications by Heijungs et al. (1992) and Fava et al. (1993) land use impacts were quantified using a simple area approach (e.g. m^2) . Land occupation (in “hectare years” or “m^2 years”) has been reported in many studies to be an impact indicator but it is really an inventory flow (Koellner, de Baan, et al. 2013) as it does not characterise the impacts associated with land use. Land occupation results on the inventory level cannot be directly compared between production systems or products in a meaningful way due to the heterogeneous nature of land qualities, including the distribution of biodiversity, which are not captured in this measure. As many studies have shown the land occupation inventory results are often a poor predictor of the final LCIA results (Mueller, de Baan, and Koellner 2014; Ridoutt et al. 2013).

Many approaches and methodologies have been proposed but, despite the unanimous recognition amongst the authors for the need to include land use impacts in LCA, most approaches had limitations which has limited their application on a global level. Some studies were limited to particular geographic regions so the approach could not be applied to products globally or applied to products where the supply chain had global reach (Geyer, Lindner, et al. 2010; Schryver et al. 2010; Köllner 2000; Koellner and Scholz 2006; Koellner and Scholz 2008, 2; Schmidt 2008). Other methods did not use spatially differentiated characterisation factors at a geographic scale that would lead to meaningful results when applied in different ecosystem types (Goedkoop and Spriensma 1999; Goedkoop et al. 2009).

To address these limitations a working group was formed - the LCIA of land use working group of the United Nations Environment Programme (UNEP) – Society of Environmental Toxicology and Chemistry Life Cycle Initiative (SETAC). The aim of the UNEP-SETAC working group is to “develop a practical tool to assess the impacts of land use anywhere on the globe” (LULCIA 2014).

The working group first developed a framework and key elements needed to assess the impacts of land use which was published in 2007 – Key Elements in a Framework for Land Use Impact Assessment Within LCA - (Canals et al. 2007). This framework was then further developed to produce a set of principles that can be used globally – Principles for life cycle inventories of land use on a global Scale (Koellner, Baan, et al. 2013). Finally, these principles were then refined to produce a set of guidelines - UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA - (Koellner, de Baan, et al. 2013). de Baan, Alkemade, and Koellner (2013) were the first to apply the UNEP-SETAC guidelines where they proposed a method to assess land use impacts on biodiversity on a global level using a relative species richness approach using Olson et al. (2001) biomes to represent global terrestrial ecosystems.

1.1.4 UNEP-SETAC guideline on global land use impact assessment on biodiversity and ecosystem services in LCA

(Koellner, de Baan, et al. 2013)

These guidelines summarise many of the agreed concepts raised by previous authors (Weidema and Lindeijer 2001; Köllner 2000; Udo de Haes et al. 2002; Canals et al. 2007) and represents the current level of agreement of the UNEP-SETAC land use working group. There is clear consensus on these guidelines and fundamental principles, but, as yet, there is no clear consensus on which impact assessment methods should be used to assess land use impacts on biodiversity. T. Koellner (pers. comm. March 2014) commented that there may never be consensus on which LCIA method should be used to assess land use impacts on biodiversity as the choice of method will vary depending on purposes of each study.

The following sub sections provide a brief summary of the fundamental guiding principles that are presented in the UNEP-SETAC guidelines.

1.1.5 Include impacts from land use occupation and land use transformation

Both land use occupation and land use transformation (or land use change) impacts need to be included in land use assessment studies. Land which is occupied for a purpose other than mature natural vegetation is prevented from returning to its natural state. Land use transformation quantifies the change in properties of the land from its natural state to another land use (e.g. natural vegetation to pasture).

1.1.6 Compare to potential natural vegetation

It is recommended that the natural state (or reference land situation), to which changes are compared, be based on the potential natural vegetation (PNV): mature vegetation that would be present in the absence of human intervention. This information for biomes and ecoregions is available in globally available land cover datasets (Koellner, de Baan, et al. 2013).

1.1.7 Include reversibility and permanence

The reversibility and permanence of land use changes and occupation also need to be considered. Reversibility is based on the concept that if the land is abandoned it will revert towards the natural state, or a quasi-natural state over a period referred to the regeneration or recovery time. For some ecosystems the impacts of land use change and occupation are irreversible and can lead to permanent impacts over an indefinite period of time (e.g. areas affected by salinity) (Koellner, de Baan, et al. 2013). The regeneration time is used in combination with the area of the land transformed to calculate the transformation impacts.

1.1.8 Document approach used to measure impacts on biodiversity

The impacts of land use on biodiversity can be measured in relative or absolute terms (e.g. percentage change or change in total number). Absolute measures treat all species with equal importance and are therefore recommended by the UNEP-SETAC working group (Koellner, de Baan, et al. 2013). Relative measures treat all ecosystems with equal importance. Most biological surveys record the relative rather than the absolute change in biodiversity associated with land use. Because of this data limitation the working group accept either approach provided it is clearly documented.

1.1.9 Ensure spatial data is collected at an appropriate spatial resolution

Land use inventory data should be collected for land occupation (measured in m^2.years) and land transformation (m^2). Attributes of the land use data, including the land use, land cover, and biogeographic region, should be gathered and stored in a GIS database. Particular attention must be given to the level of detail (spatial resolution or accuracy) of the spatial (mapping) data to ensure that it is suitable to meet the purposes of the assessment as defined in the goal and scope. The level of detail required will vary depending on the purpose of the assessment and whether the impacts are in the foreground or background systems with greater spatial resolution required in the former. In LCA, foreground data refers to inventory data which is collected for the parts of the life cycle which are under direct influence of the decision maker. Background data is collected for parts of the life cycle which the decision maker has indirect control and is often sourced from relevant literature or databases. Koellner, Baan, et al. (2013) provides further guidance on recommended land use classifications to use. Data used to differentiate biogeographic regions should at minimum be based on biomes, if not ecoregions, as defined by Olson et al. (2001).

1.1.10 Choose and/or develop appropriate characterisation factors

To reduce levels of uncertainty, generic characterisation factors for land use impacts should only be used to model impacts in the background system. Modelling impacts on the foreground system should be conducted using characterisation factors which have been developed specifically for the foreground system.

1.1.11 Manage uncertainty

Uncertainty should be managed using techniques commonly used in LCA including use of data quality assessment matrix, sensitivity analysis, and Monte Carlo simulations. Other statistical measures such as ANOVA and calculation of standard error can also be used to ensure the characterisation factors are suitable.

1.1.12 Understand the limitations

Guidance is given regarding use of results for decision making. As for any LCA study the quality of the results, and their suitability for use in decision support, depends on the quality of the LCI data collected and the suitability of the LCIA method applied. For example, if the aim is to compare the impacts of different production methods or different regions then LCI data should be collected (and foreground characterisation factors developed) for each production method or region which is to be compared.

For further details and equations for how to calculate land use impacts associated with occupation, transformation, and permanence the following papers are recommended: (Canals et al. 2007; Koellner, de Baan, et al. 2013; de Baan, Alkemade, and Koellner 2013)

1.2 Project objective

The aim of this paper is to review the current status of methods and approaches which can be used to assess land use impacts on biodiversity, the data required, and to make recommendations to MLA to include land use impacts on biodiversity in LCA studies.

2 Review of the methods

Many methods to assess land use impacts on biodiversity in LCA have been proposed but almost all had limitations which have prevented wide acceptance or adoption by LCA practitioners. Most of the methods reviewed as part of this study either:

- Did not follow the UNEP-SETAC working group best practice guidelines (as outlined previously in section 1.1.4);
- Could not be applied in areas outside of the regions for which they were originally developed;
- Were incomplete and required further research before they could be used for practical applications;
- Could not be used for products which have global supply chains and are therefore unlikely to be adopted by the international LCA community; or
- Had not been rigorously tested using suitable case studies.

This section of the report reviews those methods which either followed the UNEP-SETAC guidelines or those which could be applied to the livestock industry in Australia. Each of the following papers (in which the current methods are described) is reviewed in detail with focus on the following questions:

- What are the benefits and limitations of each approach;
- To what degree are each of the approaches and methodologies agreed by the international community; and
- What are the potential limitations with each approach?

2.1 Land use impacts on biodiversity in LCA: a global approach

(de Baan, Alkemade, and Koellner 2013)

This study was one of the first attempts to quantify the biodiversity impacts of land use in LCA on a global scale. The work followed the UNEP-SETAC LCIA of land use working group papers relating to guidance and LCI principles (Koellner, Baan, et al. 2013; Koellner, de Baan, et al. 2013), as summarised in section 1.1.4. The authors developed a global set of characterisation factors which can be used to quantify land use occupation impacts for a range of land use types within each biogeographic region around the world. The biogeographic regions used were biomes as defined by Olson et al. (2001).

Spatially explicit information on relative species richness, from peer-reviewed biodiversity surveys for a range of taxonomic groups, was used to quantify the change in biodiversity cause by each land use compared to the natural state. The resulting Biodiversity Damage Potential (BDP) characterisation factors provide indicators for “species diversity lost per area for a specific land cover relative to reference land cover [%]" (Koellner, de Baan, et al. 2013). Their approach focused on occupation impacts and excluded transformation and permanent impacts.

The authors concluded that these global characterisation factors could be used to approximate land use impacts on biodiversity although cautioned that uncertainty of the results was considerable and that the results could not be used to support decision making on land management practices. They also recommended that further research be carried out and that land transformation impacts be included.

2.2 Land use impact assessment of margarine

(Canals, Rigarlsford, and Sim 2013)

In this study the land use impacts of margarine produced for German and UK markets were assessed. These two products contained different ingredients that were grown in different locations. Land use impacts on both biodiversity and ecosystem services were included in the study. This case study was the first to apply the new characterisation factors developed by de Baan, Alkemade, and Koellner (2013) and they also followed the recommendations of the previous authors to develop a method to quantify land transformation impacts.

Life cycle inventory data was sourced from a comparative LCA study of margarine and butter, that had previously been conducted by Nilsson et al. (2010), supplemented with additional inventory data which was collected or researched. The characterisation factors for land occupation were based on de Baan, Alkemade, and Koellner (2013) which uses relative species richness as the biodiversity indicator. This previous study did not include the impacts of land transformation so a new method was proposed. This method uses FAO statistics (FAO 2011) to calculate the transformation impacts based on the increase in area which is used to grow each particular crop, within each country, over the last 20 year period. This approach to calculate transformation impacts has also been used in subsequent studies by Mueller, de Baan, and Koellner (2014).

The study successfully demonstrated that this method could be used to identify potential hotspots in the biodiversity impacts of the two types of margarine. The limitations of the approach are that it should not be used for comparison of management practices within each region unless sufficient inventory data is collected that enables a statistically valid comparison to be made. Also, the approach considers one aspect of biodiversity (relative species richness) and other aspects of biodiversity (genetic, population, or ecosystem diversity) are excluded. It should also be noted that measures of relative species richness do not take into account the absolute number of species within an area (abundance) so areas with high number of species (e.g. tropical rainforest) are treated the same as areas with a low number of species.

2.3 Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study

(Mueller, de Baan, and Koellner 2014)

Previous LCA studies comparing organic and conventional milk farming practices found no clear benefit of one over the other but these studies did not consider land use impacts on biodiversity. The aim of this study was to compare the biodiversity impact of organic and conventional milk farming practices on farms in Sweden. The study utilised previously published life cycle inventory data by Cederberg and Flysjö (2004) which collected inventory records on-farm. The supply chains of these farms had global reach sourcing supplementary feeds (e.g. grains) from Argentina, Brazil, and Malaysia. The method chosen to assess the biodiversity impacts of land use therefore also needed to also be applicable on a global basis. The methodology followed the UNEP-SETAC guidelines (Koellner, de Baan, et al. 2013) and further developed the methodology followed by (de Baan, Alkemade, and Koellner (2013) to ensure that the two farming practices could be compared.

The spatially differentiated characterisation factors of occupation and transformation were calculated based on the relative difference of plant species richness on agricultural land compared to natural regional references. Data on plant species richness were based on the study by Baan, Alkemade, and Koellner (2013) supplemented with peer-reviewed biodiversity data for each biome, land use type, and farming practice.

Following the proposed approach Weidema and Lindeijer (2001), weighting factors were applied to each characterisation factor to account for the differences in absolute species numbers and conservation value between ecoregions. The conservation values which were considered followed global biodiversity prioritisation concepts based on absolute species richness, irreplaceability and vulnerability. The data used to calculate the weighting factors were based on Olson et al. (2001).

This study is one of the most comprehensive conducted to date and is the first example in the literature where land use impacts on biodiversity have been compared between farming practices. The results indicated that although the organic production required double the land area compared to conventional production method the impacts on biodiversity were less than half. Milk produced from cows fed on meadows and pastures can have lower biodiversity impacts compared to those fed supplementary feeds, and therefore, clearly demonstrated the benefits of being able to compare biodiversity impacts of different farm practices. This highlights the benefits of making comparisons after an impact assessment method has been applied (i.e. Biodiversity Damage Potential) rather than simply on the inventory level (m^2.years).

The only limitation of the study is the time required to apply the method. T. Koellner (pers. comm., March 2014) commented that the literature review required an extensive search which took over six months. To apply the method to other regions and agricultural commodities further literature reviews are required to develop characterisation factors for each region and land use type.

2.4 Carbon, water and land use footprints of beef cattle production systems in southern Australia

(Ridoutt et al. 2013)

This study compared the land use impacts of several beef production methods in Southern Australia (Ridoutt et al. 2013). The method used global net primary productivity of potential biomass data (NPP_0)[2] data from Haberl et al. (2007) as an indicator of ecosystem productivity. This approach was originally described in Weidema and Lindeijer (2001). Net primary Productivity (NPP_0) is an indicator of the potential productivity of the land that was calculated using spatial data for climate, soil quality, land use, land cover, and statistics for agriculture and forestry production. The argument for using NPP_0 is that the use of high productivity land places greater pressure on global land resources than the use of low productivity land (Ridoutt et al. 2013).

Life cycle inventory data were sourced from previous studies by Ridoutt et al. (2012). Spatial data was then collected for each beef cattle production system including land use data from the Australian Government Bureau of Rural Science (DAFF 2006b). This study focused on occupation impacts.

The indicator results were expressed in the reference unit m^2 year-e, where 1 m^2 year-e represents 1 m^2 of land occupation for 1 year at the global average potential net primary productivity.

The benefits of this approach are that it is a relatively intuitive to understand and communicate. It can also be applied on a global level as global data for NPP_0 is available, although national Australian land use data was also used in the analysis so for it to be applied globally, alternative land use data would have to be used to assess supply chains with global reach. Normalisation factors are also available which enables comparison with other environmental impacts (e.g. carbon footprint, water footprint). The normalised results of the study indicated that the NPP_0 land use footprint was significantly higher than the carbon footprint or water footprint results. This highlights the need for land use impact methods to be included in LCA studies. The approach is relatively straight forward to apply and uses datasets that are freely available (Haberl et al. 2007; DAFF 2006b)

[2] (g C m2 yr^{-1})

Using net primary productivity as a surrogate for biodiversity also has limitations with many authors arguing that there is no evidence of a correlation between the two (Penman, Law, and Ximenes 2010; Ridoutt et al. 2013; Pfister et al. 2011; Haberl, Erb, and Krausmann 2013). For example, desert areas have low NPP_0 values but can have a high diversity of reptiles (Penman, Law, and Ximenes 2010).

Despite the limitations, this approach is still promising way to measure relative pressure on global land resources given the availability of global data, the relative simplicity of the approach, and given that normalisation factors are available.

This approach could not be used by MLA to make meaningful comparisons between the biodiversity impacts of different production systems as the results indicate the relative pressure on global land resources rather than the level of biodiversity impacts. For example it would not take into account whether grazing land was modified or unmodified natural vegetation (and the impact this has on biodiversity) as the method only considers the location, area, NPP_0 value, and yield of product.

2.5 Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets

(Coelho and Michelsen 2013)

This study was conducted to assess the biodiversity impacts associated with kiwifruit production in New Zealand. The approach followed the methodology of Michelsen (2007) to indirectly assess biodiversity quality using information on ecological structure rather than species level information. The measures of ecosystem structure used included ecosystem scarcity, ecosystem vulnerability, and conditions for maintaining biodiversity. This approach was chosen as it is relatively straightforward to apply and is largely based on spatial analysis of mapping data which is freely available on a global level.

This study did not follow the most recent guidance from the UNEP-SETAC LCIA of land use working group (Koellner, de Baan, et al. 2013) and instead followed the key elements from Canals et al. (2007). It only considered the occupation impacts of the kiwifruit orchards (excluding any potential impacts associated with the supply chain) and excluded transformation or permanent impacts of land use.

The locations of the land occupied by kiwifruit orchards (spatial inventory data) was taken from the New Zealand land cover database and divided into ten kiwifruit growing regions. No on-farm data was collected for the project.

The methodology calculated ecosystem scarcity and vulnerability at both the ecoregions level, as defined by Olson et al. (2001), and using ecosystem mapping data for New Zealand called Land Environments of New Zealand (LENZ). Scarcity was calculated based on the area of each ecosystem type divided by the area of the largest ecosystem type following Michelsen (2007). This same approach was applied to calculate ecosystem scarcity for both the ecoregions and the LENZ ecosystem types. The vulnerability of ecoregions was based on WWF Conservation status of terrestrial ecoregions (WWF 2014) reclassified to use a numeric classification system. The vulnerability of the LENZ ecosystem was based on nationally available statistics adjusted to account for the amount of vegetation in each ecosystem which had been lost. The “conditions for maintained biodiversity” is an index for the actual impact on biodiversity in the affected area. These values were estimated using a value for hemeroby (levels of naturalness) taken from Brentrup et al. (2002) although the same value was used for each of the kiwifruit growing regions meaning that comparisons between growing regions cannot be made based on this biodiversity quality indicator (i.e. to enable comparison between regions different values for hemeroby should be used).

Benefits of the approach are that ecosystem scarcity and vulnerability can be calculated using globally available datasets (Olson et al. 2001; WWF 2014) for the ecoregions level or using national datasets at a finer spatial resolution. This would enable comparison of these indicators for production in different regions. The approach is relatively straight forward to apply. The limitations of the approach is that it cannot be used for comparison of farm practices as using hemeroby does not enable these comparisons to be made in a robust manner.

This approach could not be used by MLA to make meaningful comparisons between production systems as the results would simply indicate the scarcity and vulnerability of the ecosystems in which the various aspects of each production system are based. For example, it would not provide insight into the biodiversity impacts of livestock grazed on unmodified natural vegetation compared to livestock fed concentrates.

2.6 Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction

(de Baan et al. 2013)

This study proposed a new approach which quantifies potential species extinction due to land use. The study developed spatially differentiated characterisation factors which can be used to calculate “total regional biodiversity depletion potential”, for a range of taxonomic groups, expressed in the units “potentially lost species per $m^2 \cdot$ year”. It follows the latest UNEP-SETAC guidelines (Koellner, de Baan, et al. 2013) and is the first method proposed which includes all three land use impact types; occupation, transformation, and permanence of the land use impact on biodiversity. The method can also be applied globally.

This approach models the impacts of land use on a range of taxonomic groups (plants, birds, amphibians, and reptiles) for each of the ecoregions as defined by (Olson et al.

2001).Species area curves, which are commonly used to predict species extinction due to habitat loss (de Baan et al. 2013) were used in combination with land occupation characterisation factors from de Baan, Alkemade, and Koellner (2013), and land use data from Land Degradation Assessment in Drylands (LADA, 1998 − 2008) and Anthromes (2000 – 2005) to calculate the characterisation factors.

The advantages of this approach is that it is relatively straight forward to apply using the characterisation factors provided and spatial information for the location of land use activities in the product life cycle. The results are relatively easily communicated as species extinction is a commonly understood concept. The method also is the first to include land use impacts associated with land occupation, transformation, and permanence as outlined in the latest UNEP-SETAC LCIA land use working group guidelines (Koellner, de Baan, et al. 2013). The method can also be used for retrospective or prospective assessments.

The limitations of the approach are the uncertainty of the characterisation factors and the broad nature of the land use classes used (agriculture, pasture, managed forests, urban area, and natural habitat) which do not enable comparison between land use management practices (e.g. organic vs. conventional agriculture). To date no case studies have yet been carried out to apply this method and test the robustness of the results. Due to these limitations we recommend that this approach not be used in isolation but could be applied and compared with other methods.

2.7 Summary

Each of the methods to assess the land use impacts on biodiversity in LCA which have been reviewed in detail are summarise below in Table 1. Please refer to chapters 4 and 5 for conclusions and recommendations regarding these methods.

Table 1 Summary of current methods use to assess land use impacts on biodiversity

Title	Reference	Methods	Benefits	Limitations	Application	Follows UNEP-SETAC Guidance
Land use impacts on biodiversity in LCA: a global approach	(de Baan, Alkemade, and Koellner 2013)	Species richness	Global characterisation factors available	Relative species richness, not for comparing management practices	Hotspot analysis	Yes
Land use impact assessment of margarine	(Canals, Rigarlsford, and Sim 2013)	Species richness	Land transformation included	Relative species richness	Hotspot analysis	Yes
Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study	(Mueller, de Baan, and Koellner 2014)	Species richness vulnerability, irreplace-ability	Comprehensive results enables comparison between management practices	Time intensive method	Hotspot analysis, Regional comparison, management practices comparison	Yes
Carbon, water and land use footprints of beef cattle production systems in southern Australia	(Ridoutt et al. 2013)	Net primary productivity NPP_0	Simplicity, Global data available	Land pressure rather than biodiversity indicator, only considers land occupation	Hotspot analysis, Regional comparison	No
Land use impacts on biodiversity from kiwifruit production in New Zealand assessed with global and national datasets	(Coelho and Michelsen 2013)	Ecosystem scarcity, ecosystem vulnerability, (hemeroby)	Simplicity, Global data available	Simplicity of results, only considers land occupation	Regional comparison	No
Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction	(de Baan et al. 2013)	Potential species extinction	Occupation, transformation, and permanent land use impacts included	Not yet tested with case studies, uncertainty	Hotspot analysis	Yes

3 Review of life cycle and spatial data

This section reviews the availability of life cycle (inventory) data and spatial data that are required for many of the methods previously discussed.

Note that this section does not include a literature review of peer-reviewed quantitative biodiversity surveys that would be required to then apply the approach based on species richness as used in Mueller, de Baan, and Koellner (2014). This review would be carried out after the method and region have been chosen.

3.1 Life cycle data

There are over 22 life cycle assessment studies that have been conducted for beef cattle and sheep in Australia which have been listed in Table 2 (Renouf and Fujita-Dimas 2013).

Table 2 Summary of Australian beef and sheep life cycle assessment studies (based on Renouf and Fujita-Dimas (2013)

Title	Products	Reference	Scope (cradle to)	Geographic Scope
Setting reporting periods, allocation methods and system boundaries for Australian agricultural life cycle assessment	Beef cattle and sheep	(SJ Eady and Ridoutt 2009)	Distribution	NSW case study (1 farm)
Assessing agricultural soil acidification and nutrient management in life cycle assessment	Beef cattle and sheep	(Gregory M. Peters et al. 2011)	Processing	NSW, VIC, WA case study (3 farms)
Southern Red Meat Production – a Life Cycle Assessment. Final Report to MLA	Beef cattle and sheep	(Gregory M. Peters et al. 2009)	Processing	NSW, VIC, WA case study (3 farms)
Red meat production in Australia: life cycle assessment and comparison with overseas studies	Beef cattle and sheep	(Gregory M. Peters et al. 2010)	Processing	NSW, VIC, WA case study (3 farms)
Accounting for water use in Australia red meat industry	Beef cattle and sheep	(Greg M. Peters et al. 2010)	Processing	NSW, VIC, WA case study (3 farms)
On-farm greenhouse gas emissions and water use: case studies in the Queensland beef industry	Beef cattle	(Sandra Eady, Viner, and MacDonnell 2011)	Primary production	QLD case study (2 farms)
A comparative analysis of on-farm greenhouse gas emissions from agricultural enterprises in south eastern Australia	Beef cattle	(Browne et al. 2011)	Primary production	VIC case study (2 farms)
Comparing Carbon and Water Footprints for Beef Cattle Production in Southern Australia	Beef cattle	(Ridoutt, Sanguansri, and Harper 2011)	Primary production	NSW case study (6 farms)
Water footprint of livestock: comparison of six geographically defined beef production system	Beef cattle	(Ridoutt et al. 2012)	Primary production	NSW case study (6 farms)
Assessing carbon, water and land use footprints for beef cattle production in southern Australia	Beef cattle	(Ridoutt et al. 2013)	Primary production	NSW case study (6 farms)

Title	Products	Reference	Scope (cradle to)	Geographic Scope
Undertaking a Life Cycle Assessment for the Livestock Export Trade	Beef cattle and sheep (live export)	Eady, 2013 unpublished	Distribution	-
Northern Australian Beef Supply Chain Life Cycle Assessment	Beef cattle	(Wiedemann, Murphy, McGahan, Renouf, et al. 2013)	Consumption	QLD case study (2 farms)
Life cycle assessment of four southern beef supply chains	Beef cattle	(Wiedemann, Murphy, McGahan, Bonner, et al. 2013)	Consumption	NSW, VIC case study (4 farms)
The Environmental Intensity of Australian Beef Production from 1981 to 2011	Beef cattle	(Weidemann et al. 2013)	Consumption	Australia
Global warming contributions from wheat, sheep meat and wool production in Victoria, Australia - a life cycle assessment	Sheep, co-produced with wheat	(Biswas et al. 2010)	Primary production	VIC case study (1 farm)
A comparative analysis of on-farm greenhouse gas emissions from agricultural enterprises in south eastern Australia	Sheep (for lamb and wool)	(Browne et al. 2011)		VIC case study (2 farms)
Life Cycle Assessment of Victorian Prime Lamb	Sheep	(Fisher, Reynard, et al. 2011)		VIC
Life cycle assessment of different prime lamb systems	Sheep	(Fisher, Williams, et al. 2011)		VIC
Modelling complex agricultural systems with multiple food and fibre co-products for life cycle assessment	Sheep, co-produced with grains	(Sandra Eady, Carre, and Grant 2012)	Primary production	WA case study (1 farm)
Meat consumption and water scarcity: beware of generalizations	Sheep (for meat)	(Ridoutt and Sanguansri 2012)	Retail	VIC case study (1 farm)

The majority of these studies have focused on the New South Wales, Victorian, and Queensland regions. These assessments have all been case studies of between one to six farms at a time. They also focused mainly on carbon and water footprints with the exceptions being Wiedemann, Murphy, McGahan, Renouf, et al. (2013); Wiedemann, Murphy, McGahan, Bonner, et al. (2013); and Ridoutt et al. (2013) which included metrics for land occupation or land footprinting. These three recent studies (published in 2013) could be used as a basis to assess the land use impacts on biodiversity in Australian supply chains. The availability of data for each of these studies will be briefly summarised below.

3.1.1 Life Cycle Assessment of four southern beef supply chains

(Wiedemann, Murphy, McGahan, Bonner, et al. 2013)

This study investigated the environmental impacts of different beef supply chains using grass-fed and grain-fed production systems. The study used data from four farms and three feedlots in southern Australia (New South Wales and Victoria). Several environmental impact indicators and inventory results were included in the study including land occupation. Land occupation was recorded as a measure of resource utilisation, in m^2.years, for three classes:

1. Arable land – land occupation for grain cropping, forage cropping or grazing during a pasture ley;
2. Modified non-arable grazing land – land that was cleared and in some cases sown with legume and grass species and fertilised with super phosphate; and
3. Unmodified non-arable grazing land – land that is utilised for grazing with minimal disturbance of the natural vegetation, no added pasture species, and no added fertiliser.

The area of land in each of the above classes was calculated by information provided by the farmers (confidential), GIS software and aerial imagery, summarised and reported for arable land and non-arable land occupied over one year (m^2.year).

Potential impacts of land occupation, such as impacts on biodiversity or ecosystem services, were not included in this study. Land use change (transformation) was also not included in the scope of the study both for land use nor global warming potential.

The inventory data collected during this study could form the basis for a study to assess land use impacts on biodiversity although additional research will be required to collect all relevant supply chain data to also include transformational impacts of land use.

3.1.2 Northern Australian Beef Supply Chain Life Cycle Assessment

(Wiedemann, Murphy, McGahan, Renouf, et al. 2013)

FSA Consulting also completed a LCA study focussing on northern Australian beef supply chains (Wiedemann, Murphy, McGahan, Renouf, et al. 2013). The study included two supply chains: grass fed bullocks in north-east Queensland and grain finished beef from south-west Queensland. This study included land occupation (inventory) results for two classes: arable land occupation and non-arable land occupation. The results indicated that arable land occupation was higher for the grain finished beef but non-arable land occupation was significantly higher than for grass fed beef (on rangelands) due to the lower stocking densities. It should be noted that land occupation reports results on the inventory level and does not indicate the magnitude of the impacts which result from the land occupation.

Generally, production systems with higher efficiency were associated with improved environmental performance (i.e. lower greenhouse gas emissions). However, conclusions cannot be made as to the environmental impacts of land occupation and it was recommended that further research be undertaken to assess the impacts of land use.

The inclusion of on farm inventory data from broad scale (rangeland) beef cattle production systems in this study could be interesting to assess from a biodiversity impacts perspective, and be a valuable source of inventory data.

3.1.3 Carbon, Water and Land Use Footprints of Beef Cattle Production Systems in Southern Australia

(Ridoutt et al. 2013)

Land use impacts were assessed for beef produced in Southern Australia using a proposed “resource-based” approach to land use footprinting (Ridoutt et al. 2013). The study compared six beef cattle production systems in New South Wales ranging in farm practice (grass and feedlot finishing), product (12-15 month old yearling cattle to 24-36 month old heavy steers), climate (high-rainfall coastal to semi-arid inland) and water stress. Life cycle inventory data was sourced from NSW government farm enterprise budgets rather than on-farm.

Spatial based inventory data for each beef cattle production system was compiled and classified into four classes: unimproved pasture, non-irrigated improved pasture, irrigated improved pasture and cropland. Due to relative minor contribution or lack of background data land use associated with infrastructure and resource extraction were excluded.

The conclusions of the study were that application of land occupation results on the inventory level are limited but are more meaningful when combined with spatially derived characterisation factors that include attributes on the quality of the land, such as NPP_0. Importantly there was also no correlation between the results for carbon footprint, water stress footprint and land use footprint which highlights the need for an indicator (and agreed method) for land use impacts on biodiversity.

This study may be suitable source of inventory data for a study to assess other methods of quantifying land use impacts on biodiversity especially since the carbon footprint, water stress footprint, and land use footprint (based on NPP_0) has been completed for six different beef cattle production systems in Australia.

3.2 Spatial data

The UNEP-SETAC guidelines (Koellner, de Baan, et al. 2013) and numerous previous authors (Canals et al. 2007; Ridoutt et al. 2013; Coelho and Michelsen 2013; Geyer, Stoms, et al. 2010; Geyer, Lindner, et al. 2010; Swan 1998) have highlighted the need for biogeographic data that describes various properties of the land. This type of information is often referred to as spatial datasets, GIS layers, or themes. These spatial datasets can store data in a range of data types (point, line, polygon, or grid), scale of resolution (e.g. 1:100,000), and spatial coverage (e.g. global, national, regional, local). Generally the more accurate the data the better but this higher level of accuracy often means larger, often unwieldy file sizes, which can further complicate the data analysis process. In this chapter we list the key attributes of the spatial datasets that are used or could be used to assess land use impacts on biodiversity on a global or national (Australian) scale.

3.2.1 Global geographic coverage

The global spatial data that are required for the methodologies previously discussed are described in the following section.

3.2.2 Global biomes and ecoregions

Olson et al. (2001) produced a map of the world's biodiversity dividing it into 14 biomes and 867 ecoregions. The research was based on regional analysis of biodiversity across the globe and further developed previous biogeographic mapping in consultation with over 1000 biogeographers, taxonomists, conservation biologists, and ecologists from around the world (Olson et al. 2001). The ecoregions were defined based on parameters including: species richness; endemism; higher taxonomic uniqueness; extraordinary ecological or evolutionary phenomena; and global rarity of the major habitat type. The data was published in 2001. Within Australia there are 7 biomes and 37 ecoregions.

Biomes are regions of the planet that can be classified based on their climate, fauna, and flora. They define an ecoregion as "large unit of land or water containing a geographically distinct assemblage of species, natural communities, and environmental conditions" (Olson et al. 2001). Biome and ecoregion spatial data and maps is available on the WWF website (WWF 2014).

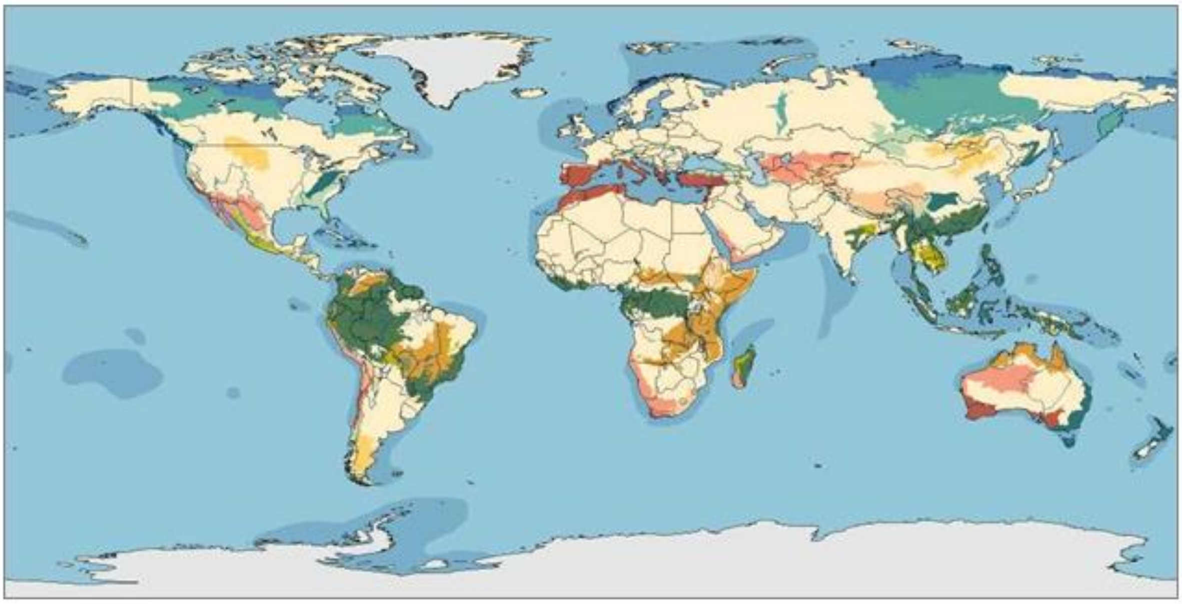

Figure 1 Global biomes (WWF 2014)

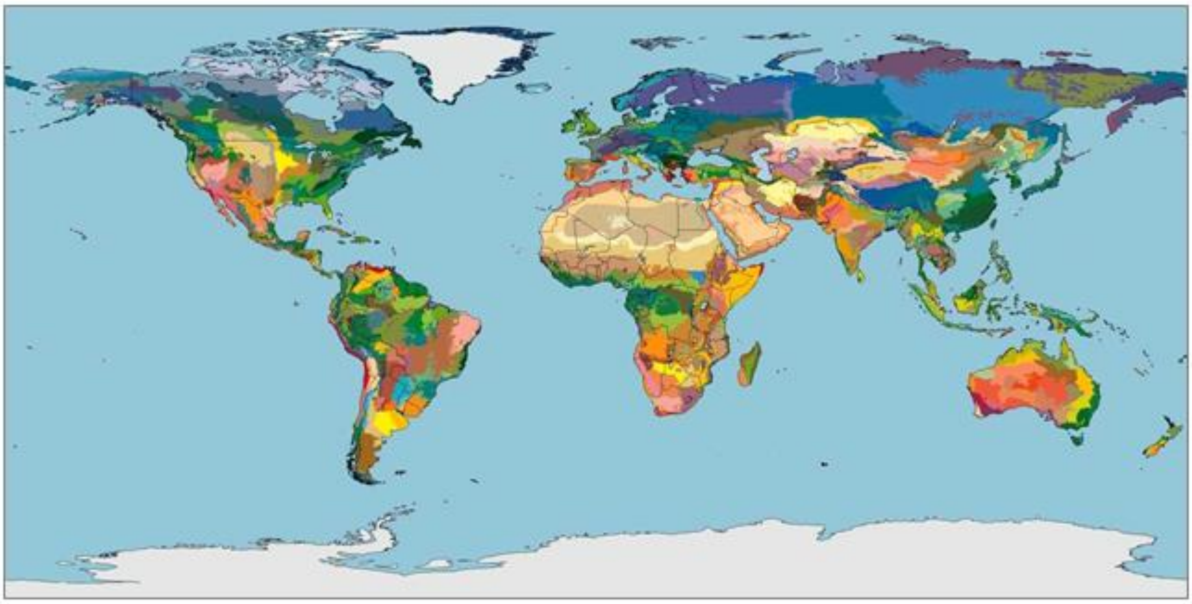

Figure 2 Global terrestrial ecoregions (WWF 2014)

3.2.3 Conservation status of terrestrial ecoregions

Not all of the world's biomes and ecoregions are equally protected. Hoekstra et al. (2004) compared the extent of each of the world biomes and ecoregions against Global Land Cover 2000 dataset (GLC 2000) and how much of each habitat was in protected areas (WDPA 2014) to produce global maps of crisis ecosystems. The conservation status is rated as: critical or endangered (red); vulnerable (orange); to relatively stable or intact (green). This data can be used as a measure of ecosystem vulnerability as used in (Coelho and Michelsen (2013). The data is available through the WWF website (WWF 2014).

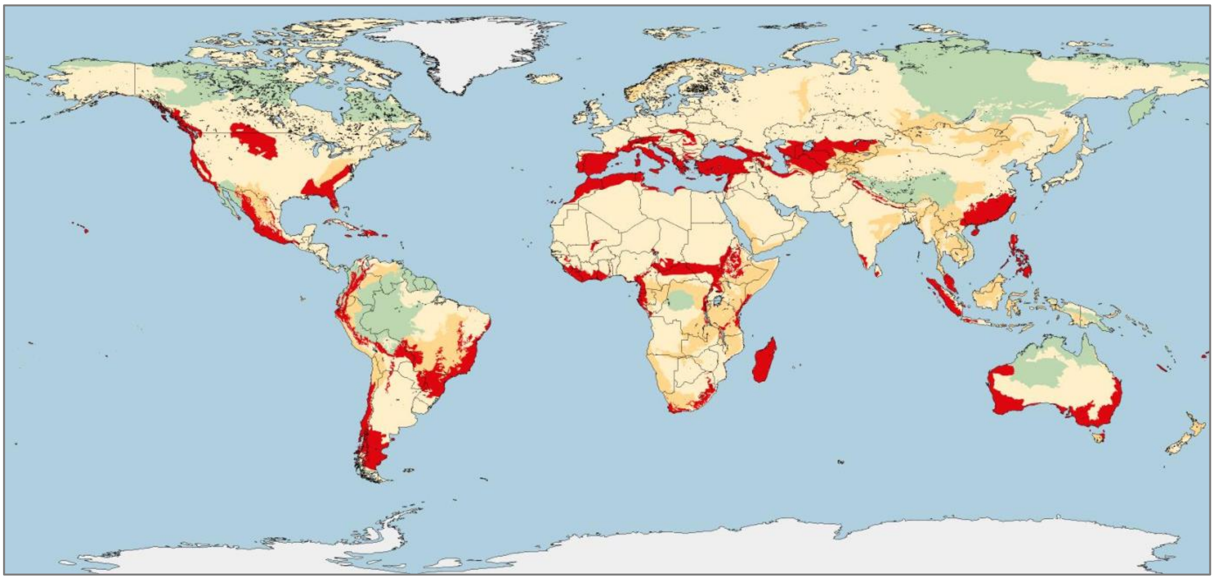

Figure 3 Conservation status of terrestrial ecoregions (WWF 2014)

3.2.4 Global land cover data (GLC2000)

The global land cover database was the first dataset that gave a detailed picture of land cover on the earth. The data was produced from daily satellite imagery (collected over a 14 month period from November 1999 to December 2000). The data was produced by 30 regionally based research teams to ensure the data was as accurate as possible. The data resolution is 1km^2 per pixel. The data was produced following the FAO Land Cover Classification System (LCCS) standard (see Appendix 2 GLC2000 Global land cover classes). The data is available through the Global Land Cover 2000 Database website (European Commission, Joint Research Centre 2003).

There is often confusion surrounding the terms “land cover” and “land use”. Land use refers to the purpose to which the land is assigned and how the land resources are used (DAFF 2006a). This data cannot always be captured by remote sensing and is often sourced from relevant surveys and statistics. Land cover describes the physical surface of the earth for example whether it is vegetation, soil, rock or water. Land cover data is often captured by remote sensing (DAFF 2006a).

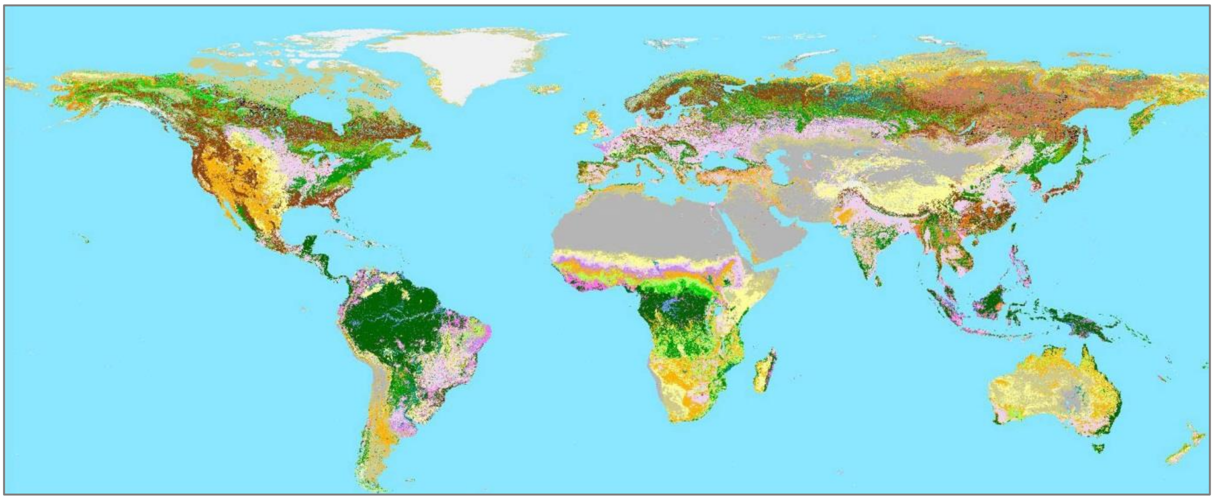

Figure 4 Global land cover data 2000 (European Commission, Joint Research Centre 2003)

3.2.5 World Database on Protected Areas

The World Database on Protected Areas (WDPA) is the only comprehensive global inventory on the world's protected areas. The WDPA is a joint venture between United Nations Environment Programme World Conservation Monitoring Centre (UNEP-WCMC) and the International Union for Conservation of Nature (IUCN) World Commission on Protected Areas. The dataset is based on national protected areas databases that are standardised using the IUCN protected areas definitions.

The spatial resolution of the data varies depending on the original scale of capture but it represents the best available data. The database is updated on an annual basis. Data is available from the WDPA website (WDPA 2014). The licence agreement for this data requires permission before being used for commercial purposes.

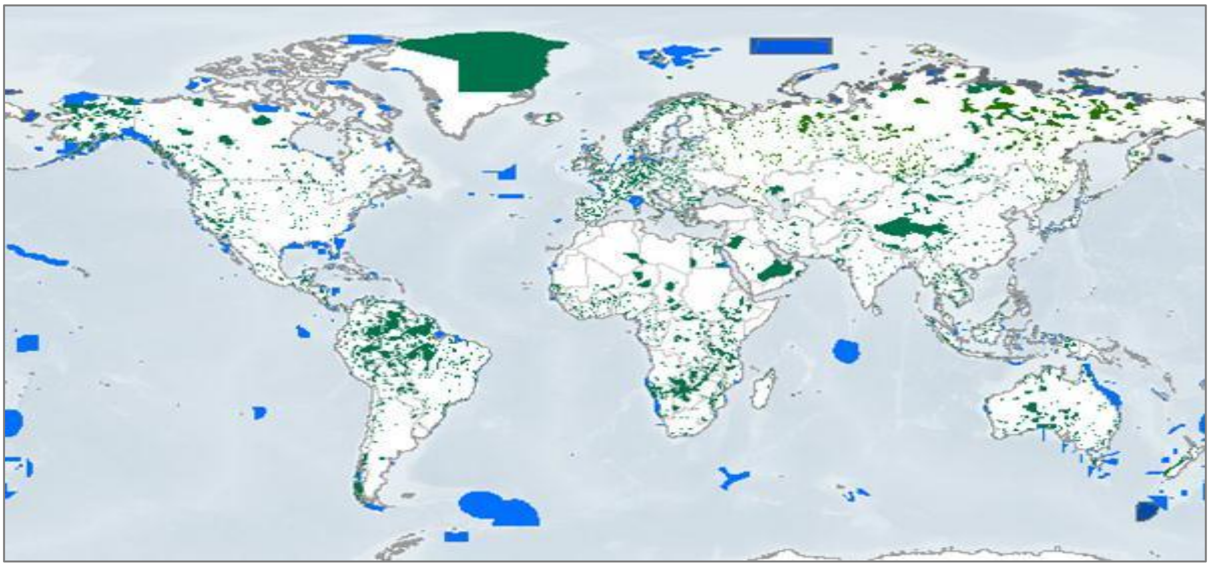

Figure 5 World Database on Protected Areas 2013 (green areas are terrestrial protected areas and blue areas are marine protected areas)

3.2.6 Global Human Appropriation of Net Primary Productivity (HANPP)

Haberl et al. (2007) produced global data which quantifies the human appropriation of net primary production (HANPP) in a spatially explicit aggregated indicator that includes both the amount of area used by humans and the intensity of the land use. It is described as being a "measure of the human domination of the biosphere" (Haberl et al. 2007). HANPP is a relative measure of how human activities (land conversion and biomass harvesting) affect the energy flows (net primary productivity) that would otherwise be available to ecosystems.

The datasets are produced using a range of sources including spatial data for temperature, precipitation, soil quality, land use, land cover, and statistics for agriculture and forestry production. The data was based on information collected for the year 2000 and the spatial resolution is 5 minute grid, which is approximately 100 km^2 (10 km by 10 km) pixel size at the equator (Haberl et al. 2007). Due to the coarse resolution of the national agriculture statistics used the authors recommend that it be used for global level or continental scale assessment and not for national or regional level assessments (Haberl, Erb, and Krausmann 2013).

The global HANPP datasets can be downloaded from the Institute for Social Ecology webpage (Institute for Social Ecology 2009).

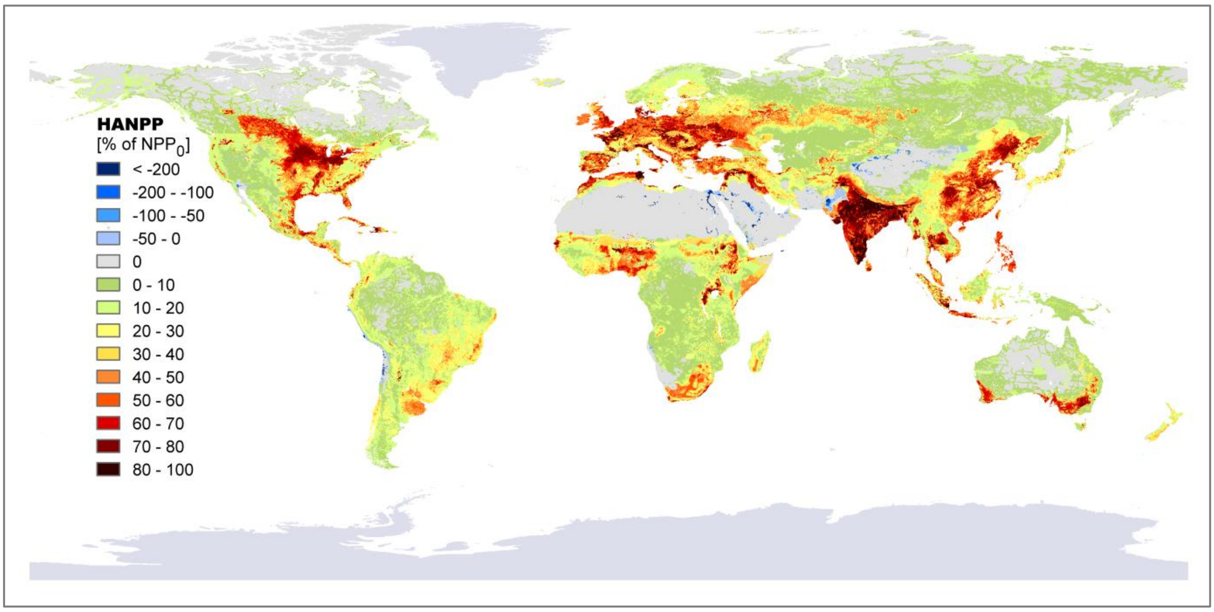

Figure 6 Global human appropriation of net primary productivity (HANPP), (Haberl et al. 2007)

Ridoutt et al. (2013) used a supplementary dataset produced as part of the HANPP series in their analysis known as net primary productivity of the vegetation that would be assumed to prevail in the absence of human land use (potential vegetation) and is abbreviated NPP_0.

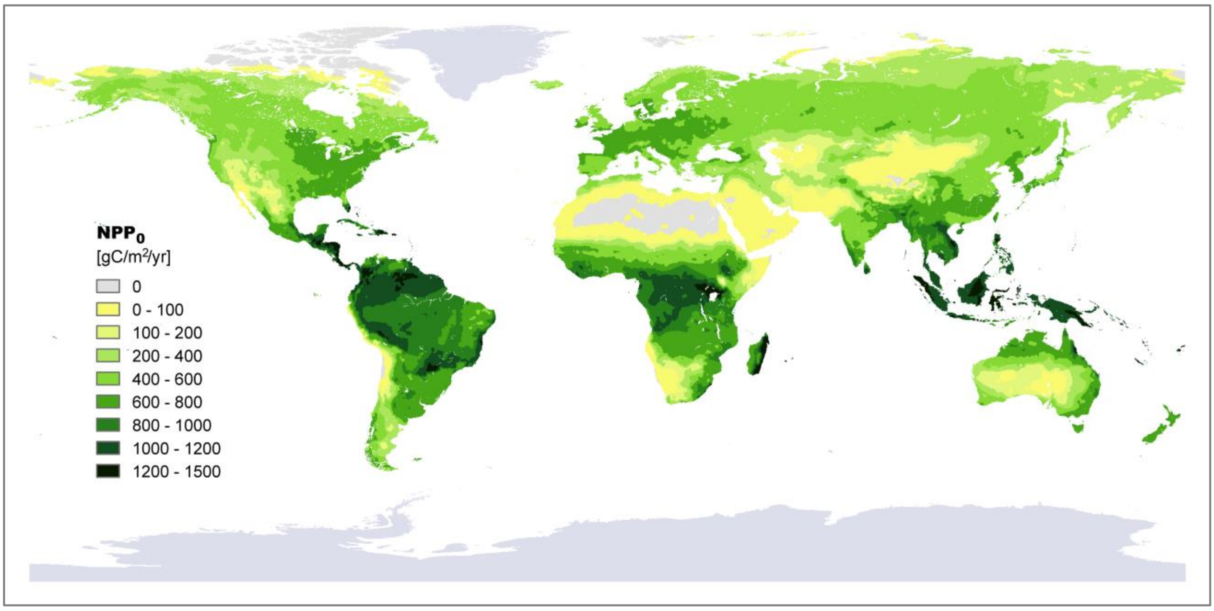

Figure 7 Global net primary productivity data (potential vegetation), NPP_0 (Haberl et al. 2007)

3.2.7 GLOBIO3

The GLOBIO3 model and data was developed to assess human-induced change in biodiversity on a global scale (Alkemade et al. 2009). The data is compiled using the cause and effect relationship between environmental drivers and biodiversity impacts. The indicator used for biodiversity is the mean abundance of original species relative to their abundance in undisturbed ecosystems. It has been used for both global and regional biodiversity assessments. The dataset is compiled using literature search of peer reviewed research on species composition in disturbed and undisturbed areas. The land cover data used as the basis for the analysis was IMAGE 2.4 (Bouwman and Kram 2006). Global Land Cover 2000 (European Commission, Joint Research Centre 2003) was also used to increase the spatial resolution from 0.5 degree cells of the IMAGE data. The GLOBIO3 data is available from the GLOBIO research team (GLOBIO Consortium 2014).

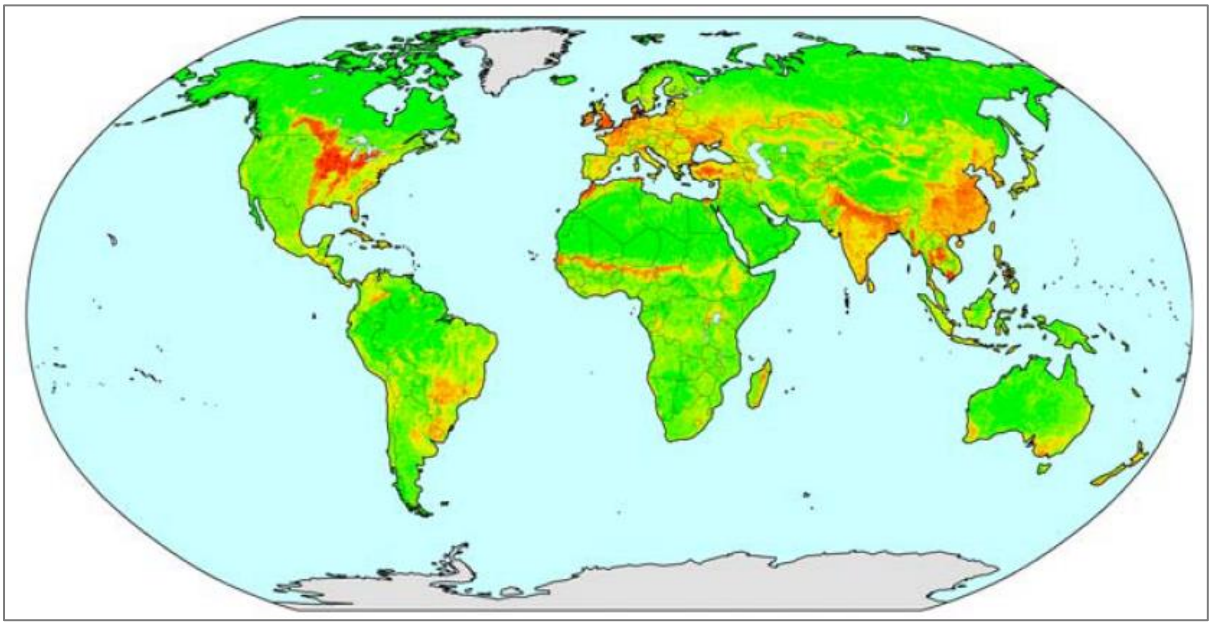

Figure 8 GloBio3 Combined relative mean species abundance of original species 2000 (Alkemade et al. 2009)

Data from the GLOBIO3 database has been used in several biodiversity impact LCA studies (de Baan, Alkemade, and Koellner 2013; Mueller, de Baan, and Koellner 2014).

3.2.8 Australian geographic coverage

When considering foreground land use impacts on biodiversity in Australia it is appropriate to use the best available data. There is a wealth of spatial data available that describe the land use, land cover, protected areas, conservation status and biodiversity within Australia. This section introduces readers to some of these dataset and databases.

3.2.9 Australian national land use data

In Australia there are two versions of land use data which have been produced on a national and catchment scale. The data on the national scale has a spatial resolution of 1: 2,500,000 (where one pixel represents approximately 1 km^2) and the catchment scale data resolution ranges from 1: 25,000 to 1: 100,000 scale. Both datasets use the same classification system, known as ALUM, to ensure national consistency. The classification system used for the national Australian land use data is similar to the European land cover mapping classes used in the CORINE project (European Environment Agency 2013) (and those required in the methods proposed by Koellner, Baan, et al. (2013). The classification scheme and summary statistics can be found in Appendix 1.

The Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) has compiled land use data from a range of sources, including Australian Bureau of Statistics (ABS) agricultural commodity data and satellite imagery, to produce the national land use dataset. The data was published in 2010 but uses data collected for the 2005/2006 period. Historic data is also available. The catchment scale data is produced by the relevant state agencies which are coordinated by ABARES. The data is available via http://www.daff.gov.au/abares/aclump/land-use/data-download.

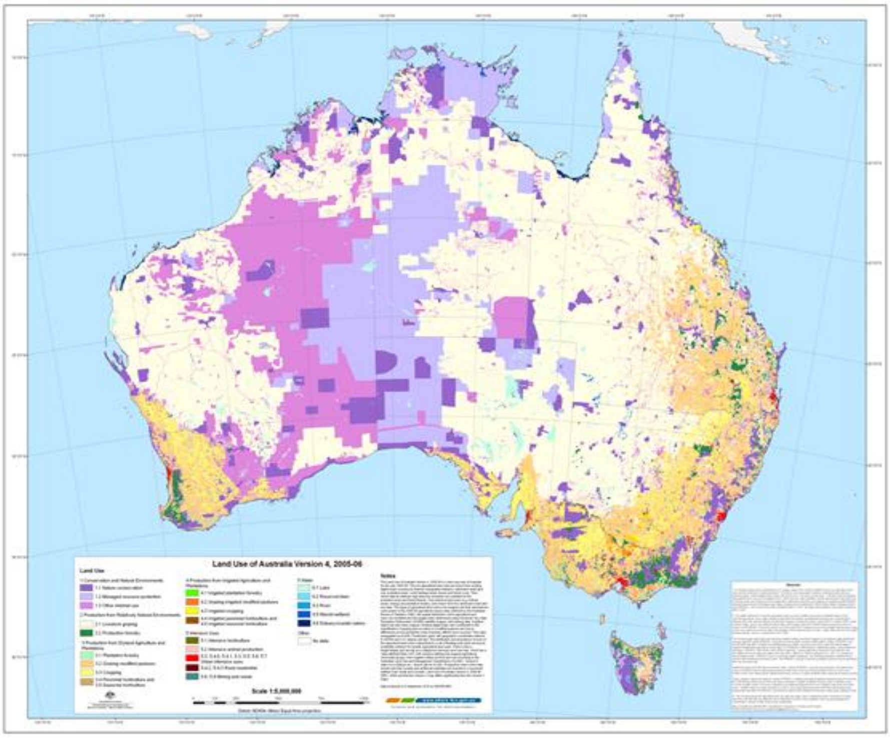

Figure 9 Australian national land use data (ABARES 2010)

3.2.10 Australian national land cover data

Australia’s first nationally consistent land cover dataset was produced in 2010 (Geoscience Australia 2011). This dataset contains information on land cover which is classified following the International Standards Organisation (ISO) land cover standard 19144-2 (2007). Australian land cover data includes 34 different ISO classes ranging from cultivated and managed land covers (crops and pastures) to natural land covers such as closed forest and sparse, open grasslands. The resolution of the data is 250m by 250m pixel size (0.6 km^2) and annual data is available from 2000 to 2008. The data is available for download from the Geoscience Australia website.

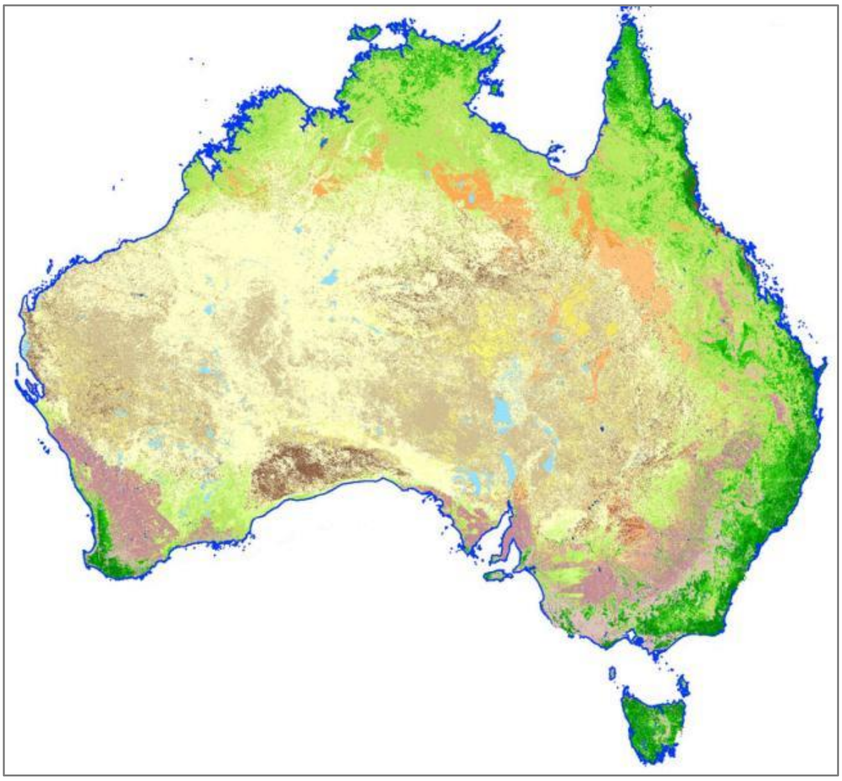

Figure 10 Australian national land cover data (Geoscience Australia 2011)

3.2.11 Collaborative Australian protected areas database

The Collaborative Australian Protected Areas Database is a compilation of data on protected areas from state and territory governments. The data is collected following the International Union for Conservation of Nature (IUCN) standards for the definition of protected areas. The data is collected every two years. Datasets are available for 1997, 2000, 2002, 2004, 2006, 2008, 2010 and 2012.

As of 2012 the database contained 10,447 terrestrial protected areas covering 118,754,264 ha (15.45 per cent) of the Australian continent. The data is available from the Department of the Environment website (Australian Government Department of the Environment 2012).

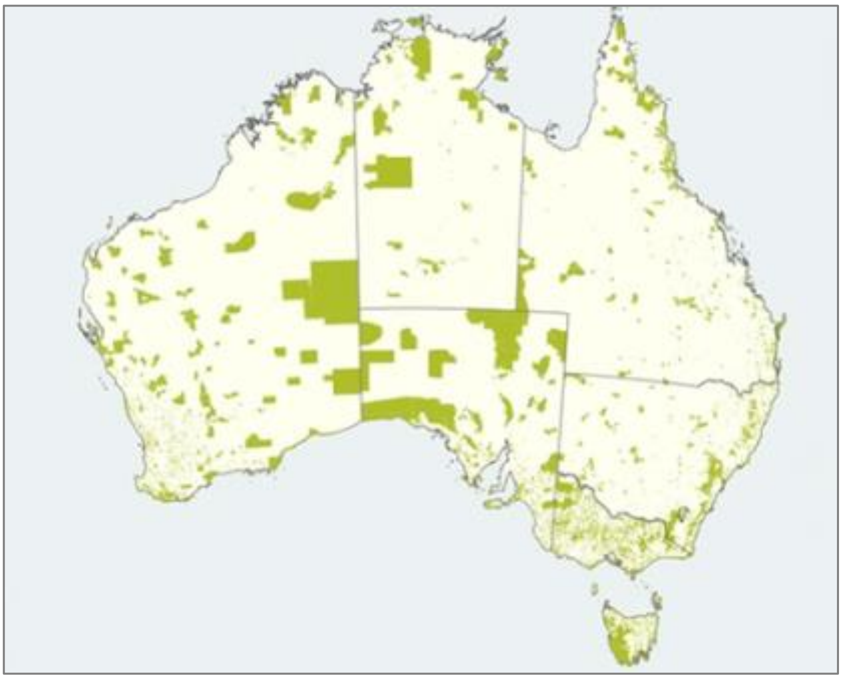

Figure 11 Collaborative Australian Protected Areas Database

3.2.12 Interim biogeographic regionalisation for Australia (IBRA)

Interim Biogeographic Regionalisation for Australia (IBRA) is a more detailed subset of the global ecoregions dataset. IBRA classifies Australia's landscapes into 89 large geographically distinct bioregions based on common climate, geology, landform, native vegetation and species information. The 89 IBRA bioregions (not to be confused with global bioregions) are further refined to form 419 subregions which are more localised and homogenous geomorphological units in each bioregion. The IBRA data is more suitable than the broader scale global bioregion or ecoregion datasets when considering impacts of land use on a finer scale (e.g. regional level).

IBRA is endorsed by all levels of government as a key tool for identifying land for conservation in Australia. The latest version of the dataset (version 7) is available from the IBRA website (Australian Government Department for the Environment 2014). The data is captured at a scale of 1: 250,000.

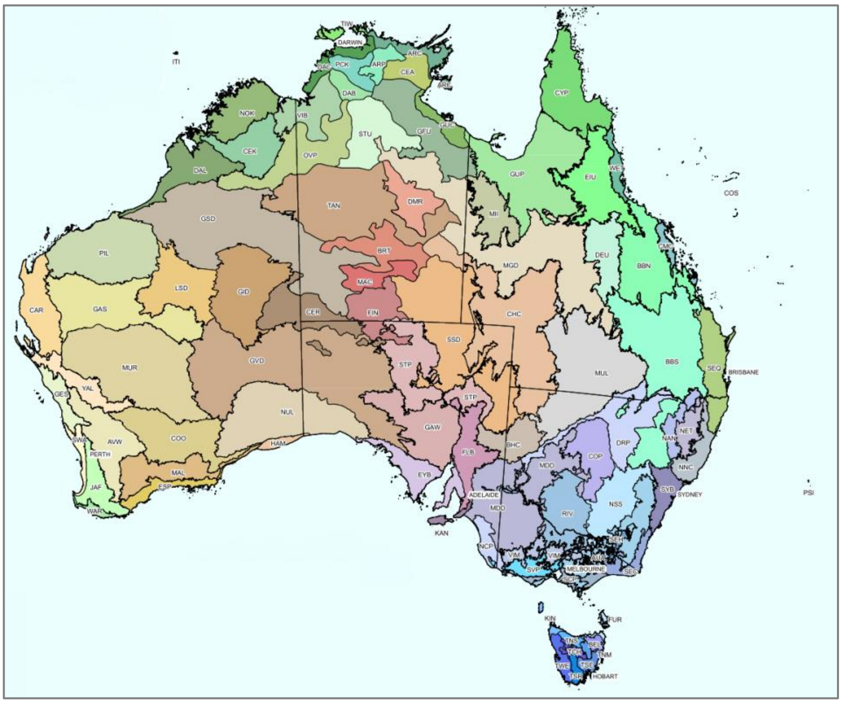

Figure 12 Interim Biogeographic Regionalisation for Australia

3.2.13 National reserve system IBRA region protection level

The protection level of the IBRA bioregions and subregions is calculated and monitored to ensure that progress towards targets for a comprehensive and adequate national reserve system is being made. All 89 Australian bioregions have some representation in the National Reserve System, with 54 bioregions having more than 10 per cent protected and 35 bioregions currently at less than 10 per cent. The data is also available from the Department of the Environment website (Australian Government Department for the Environment 2014). This data could be used as a measure of ecosystem vulnerability using a similar approach use by Coelho and Michelsen (2013) for New Zealand.

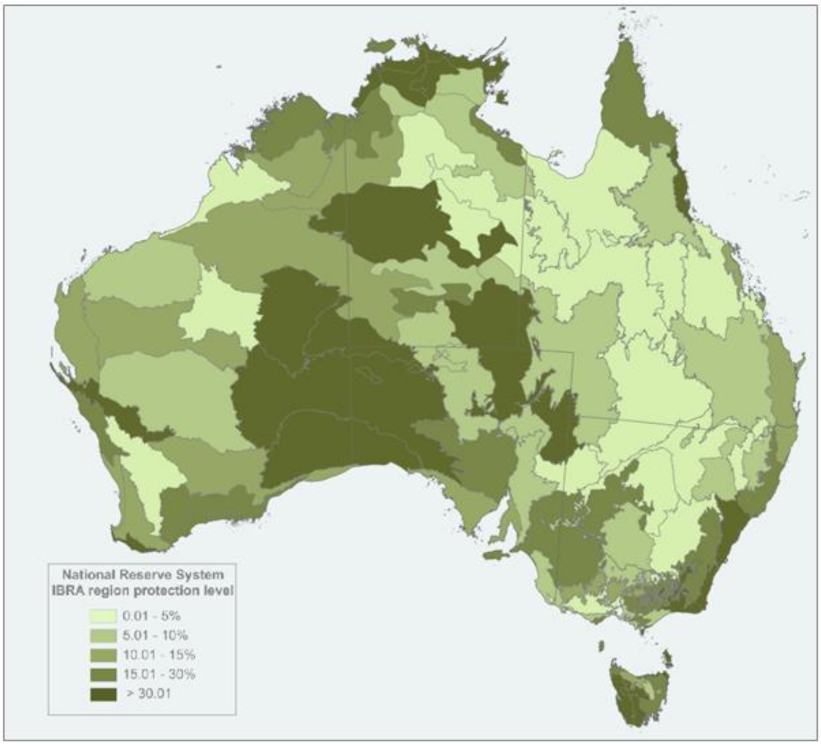

Figure 13 National reserve system IBRA region protection level

3.2.14 Atlas of Living Australia

The Atlas of Living Australia (ALA) is a national database of all of Australia's flora and fauna that aims to improve understanding of Australia's biodiversity and to assist biodiversity researchers and managers. The database is a repository for over 700 spatial datasets on Australia's biodiversity with national coverage including species richness, endemism, and threatened species. Many of these datasets could be used to assess land use impacts on biodiversity within Australia.

The database also contains a vast amount of national spatial data on climate, hydrology, soils, topography, vegetation and land use practices.

The database has been developed with links to international biodiversity projects including the European Union's Distributed Dynamic Diversity Databases for Life (4D4Life) and the Global Biodiversity Information Facility (GBIF). The database has been developed and is administered by the CSIRO. Further information can be found on the ALA website (ALA 2014)

4 Conclusions

Although the topic of assessing land use impacts on biodiversity using LCA is a complex and multidisciplinary one, simple conclusions can be made.

The first is that there is a clear need to include impacts on biodiversity in LCA to ensure that assessments are comprehensive and provide recommendations that do not lead to unintended outcomes.

Secondly, although the area has been researched for over 20 years and authors have frequently stated that there is no widely accepted method to include biodiversity impacts in LCA, significant progress has been made especially in the last two years. The UNEP-SETAC LCIA land use group has published guidelines which have been developed by experts from around the world, and represents the current consensus.

All current approaches use mapping data and GIS analysis to account for the uneven spatial distribution of biodiversity (as defined by biomes or ecoregions), land uses, and land cover over the land surface of our planet. This report reviewed both global and Australian spatial data that are available to implement the biodiversity impact assessment methods.

Robust methods are available which can be used for regions throughout the world, to quantify land use impacts on biodiversity for products including those which have global supply chains. The results can be used to identify hotspots in the life cycle (areas where improvements can be made) and to make meaningful comparisons between land management practices so that the impacts on biodiversity can ultimately be reduced.

Ultimately, quantifying the environmental impacts associated with different products, production systems, and within different regions will highlight the important issues that need to be addressed for each of them in order to make improvements in environmental performance. For example, some productions systems may have lower carbon footprints than other systems but may in turn cause higher water stress or biodiversity impacts which will then need to be focused on. By including biodiversity impacts in LCA it will ensure that this important topic will not be left out of the decision making process.

5 Recommendations

Suitable methods which can be used to assess land use impacts on biodiversity have been reviewed in this report and the benefits and limitations of each have been highlighted.

There are two approaches that could be used for livestock in Australia which are described and applied within recently published studies:

1. Land use impacts on biodiversity in LCA: a global approach (de Baan, Alkemade, and Koellner 2013), as applied in the case study: Land use impact assessment of margarine(Canals, Rigarlsford, and Sim 2013); and
2. Comparing direct land use impacts on biodiversity of conventional and organic milk—based on a Swedish case study (Mueller, de Baan, and Koellner 2014).

Both methods can be applied globally and can be used to identify biodiversity impact hotspots in the life cycle. The important difference between the two approaches is that the second approach can also be used to make comparison between farming practices, but it is a more time intensive approach to apply. The second approach is also the most comprehensive approach to assess biodiversity impacts of land use and includes measures of biodiversity on both the species and ecosystem levels.

It is recommended to proceed using an approach which follows the LCA framework (ISO 2006):

5.1 Conduct a goal and scope workshop

The aim of this workshop would be to discuss the findings of this review and to choose a particular project that could be used to apply the recommended methods.

It is recommended to use a previous LCA study as a basis for a project to assess land use impacts on biodiversity because it provides additional context, can be used as a source of life cycle inventory data, and results can be compared between other impact categories (and/or inventories) which were previously reported (e.g. land occupation, carbon footprint, water stress index). This approach has been used by many studies (Mueller, de Baan, and Koellner 2014; Ridoutt et al. 2013; Canals, Rigarlsford, and Sim 2013).

It is recommended that the biodiversity impacts of beef supply chains be used following either of the more recently published studies (Wiedemann, Murphy, McGahan, Bonner, et al. 2013; Wiedemann, Murphy, McGahan, Renouf, et al. 2013; Ridoutt et al. 2013).

5.2 Inventory analysis

Once the goal of the project has been defined the next stage is to collect relevant inventory data required to fulfil the goals of the study. This would include collecting typical life cycle inventory data (e.g. yield, feed composition, feed origin) and would also require the compilation of relevant spatial data for the products life cycle (location of land impacts, area, land use, land cover, biome, ecoregions, etc.).

Once the location of the land impacted by the supply chain is known then a detailed search for biological survey data, which is required by the different biodiversity impact methods, can be carried out. Based on the results of this search the final method (or methods) can be chosen and applied.

5.3 Impact assessment

In this stage the LCIA characterisation factors are developed and applied for one or more impact assessment methods. It is recommended that several impact assessment methods be applied so that the results of each approach can be compared to gain greater insight into the potential impacts.

5.4 Interpretation, reporting, and recommendations

The final stage is the interpretation, comparison of results, and development of recommendations. If several impact methods are chosen their results can be compared to highlight the differences and findings from each approach. Recommendations would be made identifying the hotspots and, depending on the impact methods chosen, best practice to minimise impacts on biodiversity.

6 References

ABARES. 2010. “Land Use of Australia, Version 4, 2005/2006 (September 2010 Release).” October. http://data.daff.gov.au/anrdl/metadata_files/pa_luav4g9abl07811a00.xml.

ALA. 2014. “Atlas of Living Australia.” http://www.ala.org.au/.

Alkemade, Rob, Mark Oorschot, Lera Miles, Christian Nellemann, Michel Bakkenes, and Ben ten Brink. 2009. “GLOBIO3: A Framework to Investigate Options for Reducing Global Terrestrial Biodiversity Loss.” *Ecosystems* 12 (3): 374–90. doi:10.1007/s10021-009-9229-5.

Australian Government Department for the Environment. 2014. “The National Reserve System (NRS) - Australia’s Bioregions IBRA).” http://www.environment.gov.au/topics/land/national-reserve-system/science-maps-and-data/australias-bioregions-ibra.

Australian Government Department of the Environment. 2012. “The National Reserve System (NRS) - Protected Area Information (CAPAD).” http://www.environment.gov.au/topics/land/nrs/science-maps-and-data/capad.

Biswas, Wahidul K., John Graham, Kevin Kelly, and Michele B. John. 2010. “Global Warming Contributions from Wheat, Sheep Meat and Wool Production in Victoria, Australia – a Life Cycle Assessment.” *Journal of Cleaner Production* 18 (14): 1386–92. doi:10.1016/j.jclepro.2010.05.003.

Bouwman, A.F, and T Kram. 2006. *Integrated Modelling of Global Environmental Change: An Overview of IMAGE 2.4*. Bilthoven: Netherlands Environmental Assessment Agency.

Brentrup, Frank, Jürgen Küsters, Joachim Lammel, and Hermann Kuhlmann. 2002. “Life Cycle Impact Assessment of Land Use Based on the Hemeroby Concept.” *The International Journal of Life Cycle Assessment* 7 (6): 339–48. doi:10.1007/BF02978681.

Browne, Natalie A., Richard J. Eckard, Ralph Behrendt, and Ross S. Kingwell. 2011. “A Comparative Analysis of on-Farm Greenhouse Gas Emissions from Agricultural Enterprises in South Eastern Australia.” *Animal Feed Science and Technology* 166–167 (June): 641–52. doi:10.1016/j.anifeedsci.2011.04.045.

Canals, Llorenç Milà i, Christian Bauer, Jochen Depestele, Alain Dubreuil, Ruth Freiermuth Knuchel, Gérard Gaillard, Ottar Michelsen, Ruedi Müller-Wenk, and Bernt Rydgren. 2007. “Key Elements in a Framework for Land Use Impact Assessment Within LCA.” *The International Journal of Life Cycle Assessment* 12 (1): 5–15. doi:10.1065/lca2006.05.250.

Canals, Llorenç Milà i, Giles Rigarlsford, and Sarah Sim. 2013. “Land Use Impact Assessment of Margarine.” *The International Journal of Life Cycle Assessment* 18 (6): 1265–77. doi:10.1007/s11367-012-0380-4.

Cederberg, Christel, and Anna Flysjö. 2004. “Life Cycle Inventory of 23 Dairy Farms in South-Western Sweden.” http://www.sik.se/archive/pdf-filer-katalog/SR728(1).pdf.

Coelho, Carla R. V., and Ottar Michelsen. 2013. “Land Use Impacts on Biodiversity from Kiwifruit Production in New Zealand Assessed with Global and National Datasets.” *The International Journal of Life Cycle Assessment* 19 (2): 285–96. doi:10.1007/s11367-013-0628-7.

DAFF. 2006a. “Land Use in Australia – At a Glance.” Australian Collaborative Land Use and Management Program (ACLUMP). http://data.daff.gov.au/brs/landuse/docs/Land_use_in_Australia_at_a_glance_2006.pdf.

———. 2006b. “Land Use of Australia, Version 3.” Australian Government Department of Agriculture. http://www.daff.gov.au/abares/pages/aclump/land-use/data-download.aspx.

De Baan, Laura, Rob Alkemade, and Thomas Koellner. 2013. “Land Use Impacts on Biodiversity in LCA: A Global Approach.” *The International Journal of Life Cycle Assessment* 18 (6): 1216–30. doi:10.1007/s11367-012-0412-0.

De Baan, Laura, Christopher L. Mutel, Michael Curran, Stefanie Hellweg, and Thomas Koellner. 2013. “Land Use in Life Cycle Assessment: Global Characterization Factors Based on Regional and Global Potential Species Extinction.” *Environmental Science & Technology* 47 (16): 9281–90. doi:10.1021/es400592q.

Eady, Sandra, Andrew Carre, and Tim Grant. 2012. “Life Cycle Assessment Modelling of Complex Agricultural Systems with Multiple Food and Fibre Co-Products.” *Journal of Cleaner Production* 28 (June): 143–49. doi:10.1016/j.jclepro.2011.10.005.

Eady, Sandra, James Viner, and Justin MacDonnell. 2011. "On-Farm Greenhouse Gas Emissions and Water Use: Case Studies in the Queensland Beef Industry." *Animal Production Science* 51 (8): 667–81.

Eady, SJ, and Brad Ridoutt. 2009. "Setting Reporting Periods, Allocation Methods and System Boundaries for Australian Agricultural Life Cycle Assessment." In *Proceedings of the 6th Australian Conference on Life Cycle Assessment–Sustainability Tools for a New Climate*.

European Commission, Joint Research Centre. 2003. "Global Land Cover 2000 Database." http://bioval.jrc.ec.europa.eu/products/glc2000/data_access.php.

European Environment Agency. 2013. "Corine Land Cover Types - 2006." Figure. March 12. http://www.eea.europa.eu/data-and-maps/figures/corine-land-cover-types-2006.

FAO. 2009. *THE STATE OF FOOD AND AGRICULTURE 2009 - Livestock in the Balance*. Rome. http://www.fao.org/docrep/012/i0680e/i0680e00.htm.

———. 2011. "FAOSTAT." http://faostat.fao.org/.

———. 2013. *The State of Food and Agriculture 2013 - Food Systems for Better Nutrition*. Rome. http://www.fao.org/docrep/018/i3300e/i3300e00.htm.

Fava, James A, Frank Consoli, Richard Denison, Kenneth Dickson, Tim Mohin, and Bruce Vigon. 1993. *A Conceptual Framework for Life-Cycle Impact Assessment*. Society of Environmental Toxicology and Chemistry and SETAC Foundation for Environmental Education.

Finkbeiner, Matthias. 2009. "Carbon Footprinting—opportunities and Threats." *The International Journal of Life Cycle Assessment* 14 (2): 91–94. doi:10.1007/s11367-009-0064-x.

Fisher, P. D., K Reynard, O Williams, S Fravel, N Marks, and Tim Grant. 2011. *Life Cycle Assessment of Victorian Prime Lamb (internal Publication)*. Internal. Victorian Department of Primary Industries.

Fisher, P. D., R Williams, K Reynard, and N O'Halloran. 2011. *Life Cycle Assessment of Different Prime Lamb Systems*. Internal. Victorian Department of Primary Industries.

Frank Mitloehner. 2012. "Partnership on the Environmental Benchmarking of Livestock Supply Chains: Global Feed Industry Perspectives." presented at the FAO Partnership on the environmental benchmarking of livestock supply chains: Technical seminar and launch, Rome, July 4. http://www.fao.org/fileadmin/user_upload/benchmarking/docs/mitloehner_feed_industry.pdf.

Geoscience Australia. 2011. "A Comprehensive Look at Australia's Land Cover." November 16. http://www.ga.gov.au/about-us/news-media/news-2011/a-comprehensive-look-at-australias-land-cover.html#.

Geyer, Roland, Jan P. Lindner, David M. Stoms, Frank W. Davis, and Bastian Wittstock. 2010. "Coupling GIS and LCA for Biodiversity Assessments of Land Use: Part 2: Impact Assessment." *The International Journal of Life Cycle Assessment* 15 (7): 692–703. doi:10.1007/s11367-010-0199-9.

Geyer, Roland, David M. Stoms, Jan P. Lindner, Frank W. Davis, and Bastian Wittstock. 2010. "Coupling GIS and LCA for Biodiversity Assessments of Land Use: Part 1: Inventory Modeling." *The International Journal of Life Cycle Assessment* 15 (5): 454–67. doi:10.1007/s11367-010-0170-9.

GLOBIO Consortium. 2014. "GLOBIO." http://www.globio.info/.

Goedkoop, Mark, Reinout Heijungs, Mark Huijbregts, An De Schryver, Jaap Struijs, and Rosalie van Zelm. 2009. "ReCiPe 2008: A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level." *VROM–Ruimte En Milieu, Ministerie van Volkshuisvesting, Ruimtelijke Ordening En Milieubeheer, Www. Lcia-Recipe. Net*.

Goedkoop, Mark, and Renilde Spriensma. 1999. *The Eco-Indicator 99 : A Damage Oriented Method for Life Cycle Impact Assessment : Methodology Report*. Amersfoort: PRé Consultants. www.pre-sustainability.com/download/EI99_methodology_v3.pdf.

Haberl, Helmut, Karl-Heinz Erb, and Fridolin Krausmann. 2013. "Global Human Appropriation of Net Primary Production (HANPP)." http://www.eoearth.org/view/article/153031/.

Haberl, Helmut, K Heinz Erb, Fridolin Krausmann, Veronika Gaube, Alberte Bondeau, Christoph Plutzar, Simone Gingrich, Wolfgang Lucht, and Marina Fischer-Kowalski. 2007. “Quantifying and Mapping the Human Appropriation of Net Primary Production in Earth’s Terrestrial Ecosystems.” *Proceedings of the National Academy of Sciences* 104 (31): 12942–47.

Hauschild, Michael Z., Mark Goedkoop, Jeroen Guinée, Reinout Heijungs, Mark Huijbregts, Olivier Jolliet, Manuele Margni, et al. 2013. “Identifying Best Existing Practice for Characterization Modeling in Life Cycle Impact Assessment.” *The International Journal of Life Cycle Assessment* 18 (3): 683–97. doi:10.1007/s11367-012-0489-5.

Heijungs, R., J. B. Guinée, G. Huppes, R. M. Lankreijer, H. A. Udo de Haes, A. Wegener Sleeswijk, A. M. M. Ansems, et al. 1992. *Environmental Life Cycle Assessment of Products: Guide and Backgrounds (Part 2)*. Report. CML, Leiden. https://openaccess.leidenuniv.nl/handle/1887/8062.

Hoekstra, Jonathan M., Timothy M. Boucher, Taylor H. Ricketts, and Carter Roberts. 2004. “Confronting a Biome Crisis: Global Disparities of Habitat Loss and Protection: Confronting a Biome Crisis.” *Ecology Letters* 8 (1): 23–29. doi:10.1111/j.1461-0248.2004.00686.x.

Institute for Social Ecology. 2009. “UNI Klagenfurt | Global HANPP Data.” http://www.uni-klu.ac.at/socec/inhalt/1191.htm.

IPCC. 2007. *Climate Change 2007-the Physical Science Basis: Working Group I Contribution to the Fourth Assessment Report of the IPCC*. Edited by Susan Solomon. Vol. 4. Cambridge University Press.

ISO. 2006. “ISO 14040:2006 - Environmental Management - Life Cycle Assessment - Principles and Framework.” International Standard Organization (ISO), Genève, Switzerland.

Koellner, Thomas, Laura de Baan, Tabea Beck, Miguel Brandão, Barbara Civit, Mark Goedkoop, Manuele Margni, et al. 2013. “Principles for Life Cycle Inventories of Land Use on a Global Scale.” *The International Journal of Life Cycle Assessment* 18 (6): 1203–15. doi:10.1007/s11367-012-0392-0.

Koellner, Thomas, Laura de Baan, Tabea Beck, Miguel Brandão, Barbara Civit, Manuele Margni, Llorenç Milà i Canals, Rosie Saad, Danielle Maia de Souza, and Ruedi Müller-Wenk. 2013. “UNEP-SETAC Guideline on Global Land Use Impact Assessment on Biodiversity and Ecosystem Services in LCA.” *The International Journal of Life Cycle Assessment* 18 (6): 1188–1202. doi:10.1007/s11367-013-0579-z.

Koellner, Thomas, and Roland Geyer. 2013. “Global Land Use Impact Assessment on Biodiversity and Ecosystem Services in LCA.” *The International Journal of Life Cycle Assessment* 18 (6): 1185–87. doi:10.1007/s11367-013-0580-6.

Koellner, Thomas, and Roland Scholz. 2006. “Assessment of Land Use Impacts on the Natural Environment. Part 1: An Analytical Framework for Pure Land Occupation and Land Use Change (8 Pp).” *The International Journal of Life Cycle Assessment* 12 (1): 16–23. doi:10.1065/lca2006.12.292.1.

Koellner, Thomas, and Roland W. Scholz. 2008. “Assessment of Land Use Impacts on the Natural Environment, Part 2: Generic Characterization Factors for Local Species Diversity in Central Europe.” *The International Journal of Life Cycle Assessment* 13 (1): 32–48. doi:10.1065/lca2006.12.292.2.

Köllner, Thomas. 2000. “Species-Pool Effect Potentials (SPEP) as a Yardstick to Evaluate Land-Use Impacts on Biodiversity.” *Journal of Cleaner Production* 8 (4): 293–311. doi:10.1016/S0959-6526(00)00026-3.

LULCIA. 2014. “LULCIA.” Accessed January 23. http://www.pes.uni-bayreuth.de/en/research/projects/LULCIA/.

Magurran, Anne E. 2004. *Measuring Biological Diversity*. Malden, Ma: Blackwell Pub.

Mattila, Tuomas, Tuomas Helin, and Riina Antikainen. 2011. “Land Use Indicators in Life Cycle Assessment: A Case Study on Beer Production.” *The International Journal of Life Cycle Assessment* 17 (3): 277–86. doi:10.1007/s11367-011-0353-z.

Michelsen, Ottar. 2007. “Assessment of Land Use Impact on Biodiversity: Proposal of a New Methodology Exemplified with Forestry Operations in Norway.” *The International Journal of Life Cycle Assessment* 13 (1): 22–31. doi:10.1065/lca2007.04.316.

Mueller, Carina, Laura de Baan, and Thomas Koellner. 2014. "Comparing Direct Land Use Impacts on Biodiversity of Conventional and Organic Milk—based on a Swedish Case Study." *The International Journal of Life Cycle Assessment*, January, 1–17. doi:10.1007/s11367-013-0638-5.

Nilsson, Katarina, Anna Flysjö, Jennifer Davis, Sarah Sim, Nicole Unger, and Simon Bell. 2010. "Comparative Life Cycle Assessment of Margarine and Butter Consumed in the UK, Germany and France." *The International Journal of Life Cycle Assessment* 15 (9): 916–26. doi:10.1007/s11367-010-0220-3.

Olson, David M., Eric Dinerstein, Eric D. Wikramanayake, Neil D. Burgess, George V. N. Powell, Emma C. Underwood, Jennifer A. D'amico, et al. 2001. "Terrestrial Ecoregions of the World: A New Map of Life on Earth." *BioScience* 51 (11): 933–38. doi:10.1641/0006-3568(2001)051[0933:TEOTWA]2.0.CO;2.

Penman, Trent D., Brad S. Law, and Fabiano Ximenes. 2010. "A Proposal for Accounting for Biodiversity in Life Cycle Assessment." *Biodiversity and Conservation* 19 (11): 3245–54. doi:10.1007/s10531-010-9889-7.

Peters, Greg M., Stephen G. Wiedemann, Hazel V. Rowley, and Robyn W. Tucker. 2010. "Accounting for Water Use in Australian Red Meat Production." *The International Journal of Life Cycle Assessment* 15 (3): 311–20. doi:10.1007/s11367-010-0161-x.

Peters, Gregory M., Hazel V. Rowley, Robyn Tucker, Stephen Wiedemann, M Short, Matthias Schulz, and Andrew J. Feitz. 2009. *Southern Red Meat Production – a Life Cycle Assessment*. Final Report (MLA Project COMP.094). University of NSW and FSA Consulting, for Meat and Livestock Australia (MLA).

Peters, Gregory M., Hazel V. Rowley, Stephen Wiedemann, Robyn Tucker, Michael D. Short, and Matthias Schulz. 2010. "Red Meat Production in Australia: Life Cycle Assessment and Comparison with Overseas Studies." *Environmental Science & Technology* 44 (4): 1327–32. doi:10.1021/es901131e.

Peters, Gregory M., Stephen Wiedemann, Hazel V. Rowley, Robyn Tucker, Andrew J. Feitz, and Matthias Schulz. 2011. "Assessing Agricultural Soil Acidification and Nutrient Management in Life Cycle Assessment." *The International Journal of Life Cycle Assessment* 16 (5): 431–41. doi:10.1007/s11367-011-0279-5.

Renouf, Marguerite A., and Carlos Fujita-Dimas. 2013. "Application of LCA in Australia Agriculture – a Review." In . Sydney NSW Australia: Australian Life Cycle Assessment Society. http://conference.alcas.asn.au/alcasprogram/Renouf_Review_Agriculture_Paper.pdf.

Ridoutt, Bradley G., Girija Page, Kimberley Opie, Jing Huang, and William Bellotti. 2013. "Carbon, Water and Land Use Footprints of Beef Cattle Production Systems in Southern Australia." *Journal of Cleaner Production*, August. doi:10.1016/j.jclepro.2013.08.012.

Ridoutt, Bradley G., and Peerasak Sanguansri. 2012. "Meat Consumption and Water Scarcity: Beware of Generalizations." *Journal of Cleaner Production* 28: 127–33. doi:10.1016/j.jclepro.2011.10.027.

Ridoutt, Bradley G., Peerasak Sanguansri, Michael Freer, and Gregory S. Harper. 2012. "Water Footprint of Livestock: Comparison of Six Geographically Defined Beef Production Systems." *The International Journal of Life Cycle Assessment* 17 (2): 165–75. doi:10.1007/s11367-011-0346-y.

Ridoutt, Bradley G., Peerasak Sanguansri, and Gregory S. Harper. 2011. "Comparing Carbon and Water Footprints for Beef Cattle Production in Southern Australia." *Sustainability* 3 (12): 2443–55. doi:10.3390/su3122443.

Schenck, Rita C. 2001. "Land Use and Biodiversity Indicators for Life Cycle Impact Assessment." *The International Journal of Life Cycle Assessment* 6 (2): 114–17. doi:10.1007/BF02977848.

Schmidt, Jannick H. 2008. "Development of LCIA Characterisation Factors for Land Use Impacts on Biodiversity." *Journal of Cleaner Production* 16 (18): 1929–42. doi:10.1016/j.jclepro.2008.01.004.

Schryver, An M., Mark J. Goedkoop, Rob S. E. W. Leuven, and Mark A. J. Huijbregts. 2010. "Uncertainties in the Application of the Species Area Relationship for Characterisation

Factors of Land Occupation in Life Cycle Assessment.” *The International Journal of Life Cycle Assessment* 15 (7): 682–91. doi:10.1007/s11367-010-0205-2.

Swan, Goran. 1998. “Evaluation of Land Use in Life Cycle Assessment.” *Center for Environmental Assessment of Product and Material Systems (CPM), Chalmers University of Technology, Goteborg*.

Udo de Haes, H. A., G. Finnveden, M. Goedkoop, E. Hertwich, P. Hofstetter, W. Klöpffer, W. Krewitt, and E. Lindeijer. 2002. *Life Cycle Impact Assessment: Striving towards Best Practice*. Pensacola, Florida: SETAC Press.

UNEP. 2014. “What Is Biodiversity?” *World Conservation Monitoring Centre*. http://www.unep-wcmc.org/what-is-biodiversity_50.html.

United Nations. 2013. *World Population Prospects: The 2012 Revision, Highlights and Advance Tables*. ESA/P/WP.228. United Nations, Department of Economic and Social Affairs, Population Division. http://esa.un.org/unpd/wpp/Documentation/pdf/WPP2012_HIGHLIGHTS.pdf.

WDPA. 2014. “World Database on Protected Areas.” http://www.wdpa.org/Default.aspx.

Weidema, Bo P, and E Lindeijer. 2001. “Physical Impacts of Land Use in Product Life Cycle Assessment.” *Technical University of Denmark*.

Weidemann, S. G., B. H. Henry, Eugene J McGahan, Tim Grant, Caoilinn M Murphy, and G Neithe. 2013. *The Environmental Intensity of Australian Beef Production from 1981 to 2011*. FSA Consulting for Meat and Livestock Australia.

Wiedemann, S., C. Murphy, E. J. McGahan, S. L Bonner, and R. J. Davis. 2013. *Life Cycle Assessment of Four Southern Beef Supply Chains*. B.FLT.0364. FSA Consulting for Meat and Livestock Australia.

Wiedemann, S., C. Murphy, E. J. McGahan, M. A. Renouf, P. Prasad, S. L Bonner, E. N. Zadow, and B. H. Henry. 2013. *Northern Australian Beef Supply Chain Life Cycle Assessment*. B.CCH.2028. FSA Consulting for Meat and Livestock Australia.

WWF. 2014. “WWF - G200 Maps (1999-2000).” http://wwf.panda.org/about_our_earth/ecoregions/maps/.

7 Appendix 1 Australian national land use classes

Table 3 Australian national land use classes and statistics 2005/2006 (ABARES 2010)

National Land Use 2005-06 (Version 4) Summary Statistics						
Primary Land Use	**Secondary Land Use**	**Tertiary land use**	**Area (sqkm)**	**% Total**	Area 2ndry land	% Total
1. Conservation and natural environments	1.1 Nature conservation	1.1.1 Strict nature reserves	157,044	2.04%		
		1.1.2 Wilderness area	54,125	0.70%		
		1.1.3 National park	297,009	3.86%		
		1.1.4 Natural feature protection	7,977	0.10%		
		1.1.5 Habitat/species management area	17,862	0.23%		
		1.1.6 Protected landscape	8,243	0.11%		
		1.1.7 Other conserved areas	26,980	0.35%	569,240	7.41%
	1.2 Managed resource protection	1.2.0 Managed resource protection	138,713	1.80%		
		1.2.2 Surface water supply	68	0.00%		
		1.2.5 Traditional indigenous uses	876,578	11.40%	1,015,359	13.21%
	1.3 Other minimal use	1.3.0 Other minimal use	991,490	12.90%		
		1.3.1 Defence	27,281	0.35%		
		1.3.3 Remnant native cover	223,944	2.91%	1,242,715	16.17%
2. Production from relatively natural environments	2.1 Grazing natural vegetation	2.1.0 Grazing natural vegetation	3,558,785	46.30%		
	2.2 Production forestry	2.2.0 Production forestry	114,314	1.49%	3,673,099	47.78%
3. Production from dryland agriculture and plantations	3.0 Production from dyland agriculture and plantations	3.0.0 Production from dyland agriculture and plantations	616	0.01%	616	0.01%
	3.1 Plantation forestry	3.1.0 Plantation forestry	6,703	0.09%		
		3.1.1 Hardwood plantation	6,762	0.09%		
		3.1.2 Softwood plantation	10,445	0.14%	23,910	0.31%
	3.2 Grazing modified pastures	3.2.0 Grazing modified pastures	719,566	9.36%	719,566	9.36%
	3.3 Cropping	3.3.1 Cereals	207,539	2.70%		
		3.3.3 Hay & silage	17,229	0.22%		
		3.3.4 Oil seeds	9,874	0.13%		
		3.3.5 Sugar	3,453	0.04%		
		3.3.6 Cotton	629	0.01%		
		3.3.8 Legumes	16,800	0.22%	255,524	3.32%
	3.4 Perennial horticulture	3.4.0 Perennial horticulture	94	0.00%		
		3.4.1 Tree fruits	345	0.00%		
		3.4.3 Tree nuts	185	0.00%		
		3.4.4 Vine fruits	389	0.01%	1,013	0.01%
	3.5 Seasonal horticulture	3.5.4 Vegetables & herbs	79	0.00%	79	0.00%
4. Production from irrigated agriculture and plantations	4.0 Production from irrigated agriculture and plantations	4.0.0 Production from irrigated agriculture and plantations	624	0.01%	624	0.01%
	4.1 Irrigated plantation forestry	4.1.0 Irrigated plantation forestry	19	0.00%	19	0.00%
	4.2 Irrigated modified pastures	4.2.0 Irrigated modified pastures	9,387	0.12%	9,387	0.12%
	4.3 Irrigated cropping	4.3.1 Irrigated cereals	4,163	0.05%		
		4.3.3 Irrigated hay & silage	2,878	0.04%		
		4.3.4 Irrigated oil seeds	291	0.00%		
		4.3.5 Irrigated sugar	2,427	0.03%		
		4.3.6 Irrigated cotton	2,901	0.04%		
		4.3.8 Irrigated legumes	203	0.00%	12,863	0.17%
	4.4 Irrigated perennial horticulture	4.4.0 Irrigated perennial horticulture	115	0.00%		
		4.4.1 Irrigated tree fruits	1,011	0.01%		
		4.4.3 Irrigated tree nuts	180	0.00%		
		4.4.4 Irrigated vine fruits	1,785	0.02%	3,091	0.04%
	4.5 Irrigated seasonal horticulture	4.5.4 Irrigated vegetables & herbs	863	0.01%	863	0.01%
5. Intensive uses	5.0 Intensive uses	5.0.0 Intensive uses	511	0.01%		
	5.1 Intensive horticulture	5.1.0 Intensive horticulture	31	0.00%		
		5.1.1 Shadehouses	3	0.00%		
		5.1.2 Glasshouses	5	0.00%	550	0.01%
	5.2 Intensive animal production	5.2.0 Intensive animal production	252	0.00%		
		5.2.1 Dairy	2,839	0.04%		
		5.2.2 Cattle	24	0.00%		
		5.2.4 Poultry	27	0.00%		
		5.2.5 Pigs	45	0.00%		
		5.2.6 Aquaculture	103	0.00%	3,290	0.04%
	5.3 Manufacturing and industrial	5.3.0 Manufacturing and industrial	843	0.01%	843	0.01%
	5.4 Residential	5.4.0 Residential	3,925	0.05%		
		5.4.1 Urban residential	6,418	0.08%		
		5.4.2 Rural residential	9,261	0.12%		
		5.4.3 Rural living	230	0.00%	19,834	0.26%
	5.5 Services	5.5.0 Services	293	0.00%		
		5.5.1 Commercial services	321	0.00%		
		5.5.2 Public services	299	0.00%		
		5.5.3 Recreation and culture	1,537	0.02%		
		5.5.4 Defence facilities	529	0.01%		
		5.5.5 Research facilities	353	0.00%	3,332	0.04%
	5.6 Utilities	5.6.0 Utilities	150	0.00%		
		5.6.1 Electricity generation/transmission	42	0.00%		
		5.6.2 Gas treatment, storage and transmission	17	0.00%	209	0.00%
	5.7 Transport and communication	5.7.0 Transport and communication	34	0.00%		
		5.7.1 Airports/aerodromes	302	0.00%		
		5.7.2 Roads	865	0.01%		
		5.7.3 Railways	149	0.00%		
		5.7.4 Ports and navigation	221	0.00%		
		5.7.5 Navigation and communication	13	0.00%	1,584	0.02%
	5.8 Mining	5.8.0 Mining	1,230	0.02%		
		5.8.1 Mines	252	0.00%		
		5.8.2 Quarries	45	0.00%		
		5.8.3 Tailings	23	0.00%	1,550	0.02%
	5.9 Waste treatment and disposal	5.9.0 Waste treatment and disposal	62	0.00%		
		5.9.2 Landfill	11	0.00%		
		5.9.3 Solid garbage	5	0.00%		
		5.9.5 Sewage	48	0.00%	126	0.00%
6. Water	6.1 Lake	6.1.0 Lake	25,377	0.33%		
		6.1.1 Lake - conservation	61,332	0.80%	86,709	1.13%
	6.2 Reservoir/dam	6.2.0 Reservoir/dam	5,005	0.07%		
		6.2.3 Evaporation basin	382	0.00%		
		6.2.4 Effluent pond	219	0.00%	5,606	0.07%
	6.3 River	6.3.0 River	3,508	0.05%		
		6.3.1 River - conservation	1,142	0.01%	4,650	0.06%
	6.5 Marsh/wetland	6.5.0 Marsh/wetland	5,688	0.07%		
		6.5.1 Marsh/wetland - conservation	3,637	0.05%	9,325	0.12%
	6.6 Estuary/coastal water	6.6.0 Estuary/coastal water	9,435	0.12%		
		6.6.1 Estuary/coastal waters - conservation	9,893	0.13%	19,328	0.25%
No Data			2,243	0.03%	2,243	0.03%
Total			7,687,147	100.00%	7,687,147	100.00%

8 Appendix 2 GLC2000 Global land cover classes

1. Tree Cover, broadleaved, evergreen, LCCS >15% tree cover, tree height >3m
2. Tree Cover, broadleaved, deciduous, closed
3. Tree Cover, broadleaved, deciduous, open (open 15-40% tree cover)
4. Tree Cover, needle-leaved, evergreen
5. Tree Cover, needle-leaved, deciduous
6. Tree Cover, mixed leaf type
7. Tree Cover, regularly flooded, fresh water (& brackish)
8. Tree Cover, regularly flooded, saline water (daily variation of water level)
9. Mosaic: Tree cover / Other natural vegetation
10. Tree Cover, burnt
11. Shrub Cover, closed-open, evergreen
12. Shrub Cover, closed-open, deciduous
13. Herbaceous Cover, closed
14. Sparse Herbaceous or sparse Shrub Cover
15. Regularly flooded Shrub and/or Herbaceous Cover
16. Cultivated and managed areas
17. Mosaic: Cropland / Tree Cover / Other natural vegetation
18. Mosaic: Cropland / Shrub or Grass Cover
19. Bare Areas
20. Water Bodies (natural & artificial)
21. Snow and Ice (natural & artificial)
22. Artificial surfaces and associated areas