The Economic Costs of Deforestation:

Constructing an economic account of the value of lost forest ecosystem services, with a case study of northern Argentina

UN-REDD PROGRAMME

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The UN-REDD Programme is the United Nations Collaborative initiative on Reducing Emissions from Deforestation and forest Degradation (REDD) in developing countries. The Programme was launched in September 2008 to assist developing countries prepare and implement national REDD+ strategies, and builds on the convening power and expertise of the Food and Agriculture Organization of the United Nations (FAO), the United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP).

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Acronyms and Abbreviations

ACL	American Cutaneous Leishmaniasis
BCE	Before Common Era
C	Carbon
CO_2	Carbon dioxide
DRC	Democratic Republic of Congo
EKC	Environmental Kuznets Curve
EPA	Environmental Protection Agency (of the USA)
EU	European Union
FAO	Food and Agriculture Organization (of the United Nations)
GDP	Gross Domestic Product
GHGs	Greenhouse gases
GIS	Geographic Information System
GM	Genetically Modified
GNI	Gross National Income
GWe	Gigawatts electric (i.e. 1,000 million watts of electric capacity)
GWh	Gigawatt hours
HIV	Human Immunodeficiency Virus
IPCC	Intergovernmental Panel on Climate Change
MWh	Megawatt hours
NGOs	Non-Governmental Organizations
NPV	Net Present Value
NWFPs	Non-Wood Forest Products
PES	Payments for Ecosystem Services
PgC	Petagram (i.e. 1000 million tonnes, 1 gigatonne) of Carbon
PPP	Purchasing Power Parity
REDD+	Reducing Emissions from Deforestation and forest Degradation plus conservation of forest carbon stocks, sustainable management of forest and enhancement of forest carbon stocks
SCC	Social Cost of Carbon
SEEA	System of Environmental-Economic Accounting
SEEA-EEA	System of Environmental-Economic Accounting Experimental Ecosystem Accounts
SNA	System of National Accounts
TEEB	The Economics of Ecosystems and Biodiversity
TEV	Total Economic Value
UN	United Nations
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
UNCEEA	United Nations Committee of Experts on Environmental-Economic Accounting
USA	United States of America
US$	United States (of America) dollar
WAVES	Wealth Accounting and the Valuation of Ecosystem Services
WTP	Willingness to Pay

Executive Summary

The conventional economic justification for the loss of forest has been the realizable value of wood products and use of the cleared land for agriculture, infrastructure, industry and other purposes. However, this only accounts for the direct monetary values associated with deforestation. The loss of forests has a detrimental impact on the climate through carbon emissions and reduced sequestration globally, as well as on biodiversity and other ecosystem services.

In addition to the climate change consequences, clearing forests can mean a reduction or total loss in a number of ecosystem services, resulting in reduced water quality, siltation of dams, loss of Non-Wood Forest Products (NWFPs), amongst many other impacts. By placing a monetary value on the carbon stored in forests along with the other benefits that arise from forests, it is possible to estimate the cumulative economic losses arising from deforestation over a number of years. It is then possible to adjust the monetary values associated with deforestation, in order to provide a more complete view of the economic gains from past deforestation. The more holistic approach to assessing economic activity, as adopted by this study, is in line with the ambition of the UN's Experimental Ecosystem Accounting framework (part of the System of Environmental-Economic Accounting).

A case study for a UN-REDD partner country, Argentina, is used to help develop and trial a methodology for valuing lost forest ecosystem services. The analysis uses a mixture of global and national datasets and was carried out as a desk study, so does not draw definitive conclusions about Argentina. We do estimate that the agricultural gains from deforestation are at the cost of substantial lost forest ecosystem services. The largest negative impacts in terms of cost relate to the greenhouse gas emissions and to impacts on hydrological functions. If values were calculable for all the lost ecosystem services, it is possible that the overall economic outcome would be negative.

This report also makes a broader economic case for retention of forests based on an analysis of past and present global economic development. The forest transition theory is examined and it is suggested that it may be a product of particular circumstances and partial analysis of the situation (i.e. it excludes the displacement of the deforestation footprint on to other countries). A number of examples are given of countries managing to protect or expand forest cover whilst having successful economies. In addition, there are instances of developing countries expanding food production at the same time as restoring forest cover. Therefore, active engagement in reducing deforestation is preferable to waiting for countries to reach a certain stage in economic development (as transition theory would suggest). Sustainable Development Goal 15 sets the world a high bar for 2020, of halting deforestation, promoting the sustainable management of all types of forest, restoring degraded forests and substantially increasing afforestation and reforestation. Despite international commitments, achieving this goal will be a very challenging task, and the REDD+ initiative under UNFCCC is seen as integral to doing so.

Possible sources of funding for REDD+ efforts include international results-based payments, private funding and finance, including in relation to removing deforestation from agricultural supply chains, and enlarging the domestic resources for sustainable use and protection of forests by developing countries. However funded, REDD+ can assist developing countries in pursuing their broader sustainable development objectives, promoting the retention and restoration of biodiversity, ecosystem services and associated livelihoods. Countries may wish to assess the value of lost forest ecosystem services when considering implementing REDD+, following the approach illustrated here.

1. Introduction

"Companies do not clear-cut forests out of wanton destructiveness or stupidity. On the whole, they do so because market signals – influenced by subsidies, taxation, pricing and state regulation, as well as land tenure and use rights – make it a logical and profitable thing to do. It is often profitable and logical because the costs of deforestation are generally not borne by companies clearing the land for agriculture or by companies logging and selling the timber. Rather, these costs tend to fall on society, on future generations, and often, on poor households in rural areas who frequently depend on the resources and services of the forest for their daily survival and security." *-TEEB, 2010*

Significant levels of deforestation have existed across developing countries for a number of decades (sometimes as much as a century or more)[1]. The cumulative impact of this deforestation has meant that large areas of some nations no longer have significant tree cover (Goldewijk, 2001). Whilst there have been benefits arising from this deforestation, such as wood products and increased crop or livestock production (for export or domestic use), ecosystem service (see Box 1) benefits associated with these forests have been lost.

There is an argument that whilst a certain degree of deforestation is essential for the development of an economy and society in order to free up land for other important uses[2], there comes a point when the costs associated with lost ecosystem services are likely to outweigh the (sometimes short-term[3]) benefits[4]. In some instances, alternative approaches to further increase agricultural output, such as sustainable intensification, may offer a way forward that is more beneficial overall.

Box 1. Ecosystem services

The Millennium Ecosystem Assessment report defines ecosystem services as "the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment, 2005). Ecosystem services are therefore both the tangible objects harvested from nature (such as food and fibres), as well as the processes that help support human life on the planet (such as waste assimilation and carbon sequestration).

A full economic analysis (i.e. which considers Total Economic Value – TEV) is more all-encompassing than a financial analysis of values solely as they appear through markets. Whilst a financial analysis considers financial costs and income, a full economic analysis considers all values whether or not they currently have a monetary value. Whilst some of the ecosystem service benefits of forests have market values (such as some Non-Wood Forest Products, NWFPs) others do not, or are in the process of being marketized (e.g. carbon). A number of less tangible values are unlikely to be fully realized in the

[1] … and before then in developed countries.
[2] Almost four-fifths of global deforestation is due to land conversion to agriculture (Kissinger, 2012).
[3] In relation to clear-felling for wood products, or where agricultural production is abandoned after several years due to declining soil fertility.
[4] At least at some locations where forest ecosystem services are especially valuable.

market, if at all. These include cultural benefits to local populations arising from forest ecosystem services[5].

Often at present only the realizable value of forest products and use of the cleared land for agriculture are recognized. In order to correct for missing economic values (or market failure), an analysis of the value of the ecosystem service benefits that have been lost is required. A thorough assessment would use spatial analysis, identifying where ecosystem services arise and where demand for these services exists, therefore pinpointing where benefits exist. Once the volume of benefits has been quantified it is then possible to place a value on them.

A number of economic techniques can be used to determine these values. Many involve the collection of new data via surveys. However, where budgets are limited, it is possible to use existing data in order to make such estimates. These estimates are likely to be less accurate than when new data is collected (since existing data may be from a decade or more ago, or for a different area), but should give a fairly reliable order of magnitude[6]. There will also be error associated with the identification and quantification of ecosystem services, which may be greater where it is not possible to undertake detailed spatial analysis.

Many decision-makers are not fully aware of the real costs associated with deforestation, relying on only the market values associated with wood products and the output from the subsequent land use. If the full costs are better understood, it may become clear that some deforestation is ultimately hindering rather than contributing towards (especially sustainable) economic development.

One means of addressing the issue of inadequate market signals is to introduce a national or international Payment for Ecosystem Services scheme (Kumar and Thiaw, 2013). Since part of the value of standing forests (such as avoided carbon emissions) may be felt outside of the nation, quantifying these lends support to existing multi-national transfers of funds in return for maintaining these services, and may highlight the need to further extend such fund transfers.

This report includes a broad analysis of the economics of deforestation in order to make the case for a TEV approach, as well as a detailed national case study (Argentina) in order to develop and pilot a methodology that could be used. It is hoped that practitioners will adapt and improve the approach presented here, and that it will also help inform work in the related area of ecosystem accounting.

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[5] Further to this is the importance that individuals in both developing and developed nations may place on the existence of the forests in developing nations even without any intention to visit them. Some of this value is reflected through bilateral aid or donations to NGOs, but this is unlikely to reflect the full value.

[6] If the data deficiencies are corrected as far as is possible, for example up-rating past estimates to current values using relevant inflation rates.

2. Objectives

The objectives of this report are:

i. To develop a spatially-based methodology which determines the likely costs of deforestation, i.e. the overall value of ecosystem service benefits arising from forests which have been lost, and to compare this with estimates of the financial gains.
ii. To present analyses for a case study country over five decades.
iii. To explore the economics of deforestation, specifically the external costs, and briefly explore a “Green Economy” strategy of conserving forests whilst also expanding economic activity.

Whilst the case study is provided as an example, it was also an essential means of testing the feasibility of implementation of the methodology. The broader economic investigation is intended to make the case for adoption of this analytical approach by REDD+ countries, preferably using national rather than global datasets.

3. Natural capital: the nature of wealth and the wealth of nature

Wealth is a measure of our assets and the foundation upon which the generation of income is produced. Wealth includes factories and infrastructure, but also natural wealth, such as fossil fuels, minerals and living nature (plants and animals)[7]. To determine whether existing levels of income can be sustained, we need to know whether, and by how much, our assets are being liquidized to produce this income. If so, are they being replaced by an alternative asset that can produce similar levels of income in the long term? A net overall loss means that at some point in the future, income will decline. For example, rapid economic growth can accompany the development of fossil fuel reserves. However, if this wealth is used only for day-to-day consumption and not invested in productive infrastructure, then, when these fossil fuels are no longer available, income will decline.

Although widely reported, and often seen as a key indicator of policy success (Fioramonti, 2013), GDP (Gross Domestic Product) figures have only been the focus of attention for a relatively short period. GDP represents a survey of economic activity from the perspective of financial transactions: that is, the total market value of goods and services produced within a nation's borders. Thus, GDP focuses on flows of money, and hence not on changes in assets such as forest. It also ignores the costs that businesses, institutions and individuals impose upon others but for which they are not charged (externalities), such as environmental degradation - unless countering these entails market transactions in the same country[8].

There is a causal link between increases in GDP and economic development as measured by material throughput, and hence likely improvements in the standard of living of the population. Yet, economic growth can be accompanied by declines in welfare (increasing inequality, crime, unemployment and pollution increase) (Kennedy, 1968). Therefore, a number of alternatives to GDP have been proposed (Kubiszewski et al., 2013).

To help correct for changes in natural resources a new accounting system (the System of Environmental-Economic Accounting, SEEA) has been developed by the United Nations Committee of Experts on Environmental-Economic Accounting (UNCEEA). Many countries want to go further still, to include ecosystem services and natural capital that are not traded or marketed, and so are more difficult to measure. Work on the Experimental Ecosystem Accounts (SEEA-EEA), which will facilitate this, was completed by UNCEEA in 2013. Since these are experimental accounts, there is still a need for further development. Therefore, UNCEEA is encouraging the piloting of SEEA-EEA, in order to test and experiment in this new area of statistics. To this end UN Statistics Division, UN Environment Programme (TEEB Office) and Secretariat of the Convention on Biodiversity have embarked on a project to advance, test and build capacity to implement the SEEA-EEA framework. In related work, the Wealth Accounting and the Valuation of Ecosystem Services (WAVES)[9] partnership aims to

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[7] Some people also classify money as wealth, whilst others see it as just a marker for wealth (i.e. a socially accepted medium of exchange that is a means to acquire ‘real’ wealth).

[8] In such cases these harms actually appear as additions to GDP.

[9] http://www.wavespartnership.org/en

advance the implementation of the SEEA frameworks and wealth accounting that consider natural resources (e.g., via adjusted net savings) internationally.

This current study is complementary to such national accounts, as it aims to identify the cost of the historical loss of forests, based on their full value. This historical examination of the value of natural assets, and the net economic gain or loss resulting from their liquidation, could help inform deliberations on whether continuing deforestation is likely to be in the interests of the country involved. As such it is a further step in the direction of assessing the true wealth of nature.

4. The negative impacts of deforestation on the economy and society

The benefits of deforestation are clear: use of wood as a raw material or fuel source, and the freeing up of land for agriculture, infrastructure, housing, industry or some other form of development. These outputs that result (i.e. wood products and agricultural goods) are picked up in measures of economic activity (GDP). The negative impacts of deforestation can be less obvious, although just as real, and can include lost government revenues worth billions of dollars (Moestafa, 2013). Some impacts are felt soon after the deforestation occurs, but others are delayed. The impacts (and benefits) will not only materialize where the deforestation took place, but may be felt in other countries or even globally. In native or old growth forests, deforestation may be likened to an irreversible decision (Forsyth, 2000): the value of the option to delay deforestation until these benefits are more fully understood is an important economic consideration in itself.

Changes in global forest cover over time provide an indication of the quantity of carbon being released into the atmosphere as a result. FAO (2012) estimates that global forest cover stands at 4 billion hectares, and between 2000 and 2010 the world lost about 130 million hectares of forest, with net loss being 52 million hectares[10] (FAO, 2012). Carbon dioxide (CO_2) emissions from tropical deforestation and forest degradation were estimated at 2.8 ± 0.5 PgC a year on average for the first decade of this century (IPCC, 2013). When carbon taken up by recovering and newly planted forests is taken into account, a balance of around 0.9 (± 0.8) PgC per year were emitted from land-use change from 2002 to 2011 (IPCC, 2013). This equates to approximately 10% of global CO_2 emissions for the period. In addition, emissions from the burning and decomposition of drained peatlands (also often in forest), were estimated at 0.22 to 0.35 PgC per year (1997 to 2006) (IPCC, 2013). The impacts of the resulting climate change are already being felt, mainly through increased weather extremes such as heavy rains, storms, heatwaves or droughts. Other impacts range from rising sea levels to increasing impacts from pests and diseases in a warmer climate (e.g. bark beetle infestations in North America (Carroll et al. 2003)). By 2010, global economic losses from climate change were estimated at close to 1% of GDP per year (or US$0.7 trillion, in 2010 PPP) (DARA, 2012).

In addition to global climate impacts, deforestation can have local and regional climate impacts (Lewis, 2006) that will also damage the economy. For instance, deforestation can affect local climate through hydrological cycles. Trees extract groundwater through their roots and release it into the atmosphere through transpiration, and so their removal typically reduces atmospheric moisture. Brazilian meteorologist José Marengo coined the term "flying rivers" in the 1990s to describe air currents that carry water vapour rising from the Amazon (and blocked to the west by the Andes mountains) to central and southeast Brazil as well as northern Argentina. It is thought that deforestation is reducing this flow and contributing to the current drought (the worst in at least 80 years) in the area, drying up the reservoirs that supply São Paulo (South America's biggest and wealthiest city), and impacting key agricultural crops (Brasileiro, 2014).

Other negative impacts on the economy can occur through increased soil erosion leading to siltation of rivers and dams downstream (and increased dredging costs or lost output), and impacts on

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[10] These global figures can hide high regional losses.

freshwater fisheries productivity (Crafford et al., 2012). Well-managed natural forests almost always provide higher quality water than that obtained from other catchments (Dudley and Stolton, 2003). There are also health impacts, first from haze resulting from forest burning, and also as a result of the spread of novel diseases, some of which are fatal. The 2014 Ebola outbreak in West Africa is thought to have originated in recently deforested areas[11]. The World Bank (2014b) estimates that this outbreak could have a negative financial impact in the region of almost US$33 billion by the end of 2015. Previous Ebola outbreaks have been associated with harvesting of bush meat, which does not require deforestation; and so it is important not to over-simplify this link with disease.

The loss of sustainably harvested medicinal forest products can vary from those of low market value (though still important to forest-dependent people) to potentially significant economic losses at the global scale. For instance, many forests contain very high levels of biodiversity; therefore, they have the potential to provide genetic material to develop new medicines and crop varieties. Argentina has 236 native tree species, of which 10 are identified as endangered and 34 as vulnerable (Butler, 2006). Of the world's tree species, less than 1% have been described at the genetic level (FAO, 2014b), and most have not been screened for possible use to medical science. A famous success story is the development of the cancer treatment drugs vincristine and vinblastine from the Periwinkle plants of Madagascar's forest (Clark, 1999). Ongoing deforestation probably results in the permanent loss of similarly valuable drugs.

The benefits of deforestation accrue unevenly amongst the population, and research (Rodrigues et al., 2009) has suggested that the majority of the benefits for the local population are transitory in nature. Rodrigues et al., (2009) studied 286 municipalities in the Brazilian Amazon and found that deforestation leads to social and economic 'boom and bust', i.e. although welfare (income, life expectancy, etc.) rapidly improved during deforestation, this was short-lived. This was partly due to the fact that once the wave of deforestation had passed through a community it was more populated than before, and the soil was rapidly degraded (with some land being abandoned as no longer useful for agriculture), with large-scale farms using machinery and imported fertilizers and employing few people (Rodrigues et al., 2009). In the years following deforestation development indicators dropped back to the pre-deforested rates (Rodrigues et al., 2009).

The impacts of deforestation can especially affect that proportion of the economy that can be termed 'GDP of the poor'. For example, the value of forest services such as fresh water, soil nutrients and NWFPs amounted to some 57% of the income of India's rural poor (Sukhdev, 2009). Therefore, deforestation can result in negative impacts on economic development for the poorest in society, with the gains mostly being realized by a very small minority. However, the impacts of deforestation can also be felt more widely across society. Whether deforestation has been beneficial to the economy, and to what extent, will vary from country to country. The following chapter describes a methodology to undertake an economic analysis of past deforestation, and is followed by a case study for northern Argentina, to trial and demonstrate this methodology.

[11] The Guinean forest surrounding the outbreak areas has been subject to dramatic forest loss with the landscape now dominated by forest-agricultural mosaics, providing the opportunity for direct exposure to infected bats (Alexander et al., 2014).

5. Methodology

This study sets out a general approach, with options over how detailed the individual steps are, since the time and resources available to devote to such an analysis will vary among countries. The steps envisaged are as follows (in practice some may be combined):

i. Assess the general economic situation of the country.
ii. Determine forest cover changes over time.
iii. Explore the drivers of deforestation and land-use change.
iv. Estimate the quantity of wood products and agricultural output as a result of deforestation.
v. Quantify the monetary value of this output.
vi. Identification of the main ecosystem services related to forests.
vii. Estimate the quantity of the different ecosystem services lost.
viii. Quantify the monetary value of the lost services.
ix. Estimate the corrected economic gains from deforestation.
x. Compare the results of step ix with the information gathered in the first step.

Each of the steps is explained in more detail below.

i. Assess the general economic situation of the country

First, an assessment of the general economic situation existing in the country, and over the period to be examined, is required. This is in order to understand trends in GDP, land use, wood products and agricultural exports and the size of the agricultural and forestry sectors, as well as hydroelectricity production and indicators of development (health, education and incomes of the poorest, etc).

This information will be used in the final step for comparison, but will also help to determine which data is required for step vii. It can be completed in parallel with steps ii and iii. In addition to national sources of data (such as statistical offices or ministries of finance, trade and agriculture) the following may be of use:
http://www.fao.org/statistics/en/
http://stats.oecd.org/
http://www.imf.org/external/data.htm
http://unstats.un.org/unsd/economic_main.htm
http://data.worldbank.org/

Further data may also be found amongst the research literature, NGO reports, etc.

ii. Determine forest cover changes over time

This information is required to assess changes over recent decades and would ideally also give a geographical indication of where in the country the forest loss occurred. Whilst recent forest cover data will be relatively easily obtained[12], as will global or national maps of land cover/use, data for earlier decades may be more difficult to obtain. However, historical land-cover data sets do exist based on models, such as the HYDE[13] database. This was originally designed for testing and validation of the IMAGE 2 model (Goldewijk, 2001), but has been extended to provide global maps of land-use going back to 1700 (and agricultural land use back to 10 000 BCE). See Appendix B for more details on the HYDE data and its limitations.

iii. Explore the drivers of deforestation.

The reasons for deforestation vary between countries and even between regions within countries. In some places, the value of the wood products is the dominant driver (either for timber or for fuel such as charcoal), followed by a change in land use, and in other situations the wood is not harvested but burnt *in situ* so that the land can be cleared for agriculture (cropping or pasture).

Identifying the reasons for historical land-use change is an important step, in order to identify the main values associated with the deforestation, including alternative land uses and their productive outputs following forest conversion. National documents, NGO reports and academic studies may all contain relevant information.

iv. Estimate the quantity of wood products and agricultural output as a result of deforestation.

These calculations are based on the volume of wood and agricultural products produced over each decade, which can be assigned to the area of land that was deforested – in the case of wood during that decade alone, whilst for agricultural production this output will be on-going over subsequent decades (unless the land is abandoned). The quantity of wood can be assessed by using FAO data and relevant information on tree species or forest type and volumes of harvestable wood per hectare. For agriculture it will be important to define the agricultural production system and specific crop or livestock produced, as well as average yields. Whilst detailed agricultural statistics may be available it is more likely that national-level data will have to be used (e.g. see http://faostat3.fao.org/), along with historical documents, to estimate the area of different crops grown on the deforested land and average decadal yields.

v. Quantify the monetary value of this output.

A valuation of the outputs will require information on commodity prices for each decade (decade average) since deforestation. The accuracy improvements for annual over decadal prices are likely to be marginal. International commodity price information can be found at the World Bank: http://siteresources.worldbank.org/INTPROSPECTS/Resources/334934-1304428586133/pink_data_a.xlsx as well as here: http://www.indexmundi.com/commodities/.

[12] E.g. for forest cover (and forest type) data for 1990, 2000, and 2010: http://foris.fao.org/static/data/fra2010/FRA2010GlobaltablesEnJune29.xls
[13] http://www.atmos.illinois.edu/~meiyapp2/datasets.htm

Forward projections to the final year being considered (e.g. 2099) are also required (see viii for further detail).

vi. Identification of the main ecosystem services related to forests in the country.

Ecosystem services are by definition of value to people. Ecosystem services are a result of ecosystem functioning, however to be of benefit to humans some of the services, e.g. NWFPs, require a human population to be in close proximity to where the services are generated. Others, such as carbon sequestration, do not, as they impact globally. In order to demonstrate the main ecosystem services related to tropical forests, first a general classification of ecosystem services is laid out; then this is adapted for forests; and finally the services that can be realistically valued within an economics framework are identified.

A number of different ways to categorise ecosystem services have been developed, depending on the reason for the classification.

The Common International Classification of Ecosystem Services (Haines-Young & Potschin, 2013) came about from the work on environmental accounting undertaken by the European Environment Agency in support of the revised SEEA. A common international classification is needed if ecosystem accounting methods are to be developed and comparisons made between countries. In addition to the need for standardization in the context of environmental accounting, wider work on mapping and valuing ecosystem services also benefits from more systematic approaches to naming and describing ecosystem services. As a result of recent consultations, an updated version of CICES (Version 4.3) has now been proposed (Appendix I).

Drawing on this, the classification used in the present report is as follows:

Table 1: Potential forest ecosystem services and their benefits to people

ECOSYSTEM SERVICE TYPE	BENEFITS TO PEOPLE
PROVISIONING:	
Food	Game, insects, mushrooms, fruits, vegetables, spices and honey
Medicines	Traditional medicines, genetic material used to develop new pharmaceuticals
Ornamental resources	Biomass materials for handicraft and decoration
Raw materials of energy	Providing wood products (when harvested at sustainable rates)
REGULATION & MAINTENANCE:	
Biological control	Pest and disease control
Climate stability	Supporting a stable climate at global and local levels through carbon sequestration
Air quality	Absorption of air pollutants (around urban areas)
Moderation of hazards	Reducing the impact of natural hazards, such as tidal surges along coastlines
Pollination	Pollination of plant species, including neighbouring crops

Local climate	Micro climate regulation: providing shade from the sun and shelter from winds, benefiting crops and livestock
Soil retention (erosion prevention)	Improving slope stability and coastal integrity, reducing downstream siltation (in areas prone to erosion) of rivers
Water quality	Reducing chemical and sediment loads for rivers in the watershed
Water regulation	Mediation of high and low river flows (in certain circumstances)
Genetic resources	Gene pool for improving crops (in forests with high biodiversity)
CULTURAL:	
Artistic inspiration	Using nature as motifs in art, folklore, books and cultural symbols
Recreation and tourism	Physical interactions: experiencing the natural world, appreciating the scenery and enjoying outdoor activities
Scientific understanding	Intellectual interaction: using natural systems for scientific research
Spiritual	For sacred or religious purposes
Existence and bequest	Knowing that a place is being preserved (for others or future generations)

It is extremely difficult to estimate a monetary value for some of these services. Based on the above, the ten priority ecosystem services associated with forests that might be included in an economic valuation are as follows (in order of ecosystem service type, not priority):

1. Harvested NWFPs, specifically: foods, medicines, fibres, and resins
2. Sustainably harvested wood
3. Pollination of crops bordering the forest
4. Regulation of diseases (to humans)
5. Carbon retention/sequestration.
6. Reduced sedimentation through limiting soil erosion
7. Impact on water purification: clean water for drinking and fisheries
8. Impacts on water flows: floods and droughts
9. Biodiversity and landscapes as a tourism resource
10. Existence and bequest values to national and global population

And for coastal areas:

- Impact on risk reduction from mangroves as sea defences

In some locations, such as where studies already exist, it may also be possible to place a monetary value on the following:

- Regulation of pests: the net benefit to neighbouring crops of natural pest control
- Impacts on local climate: shelter and shade
- The genetic resource for crops or new products

Some of the ecosystem services identified in the main list and above are mutually exclusive for a particular area, so double-counting must be avoided. For example, removing wood for fuel reduces carbon storage. In addition, as mentioned earlier, some of the ecosystem services will not be applicable (i.e. to humans as an economic benefit) in all situations, for instance in deep impenetrable forests there may be no harvesting of NWFPs. This issue becomes critical when estimating the quantity of lost ecosystem services.

vii. Estimate the quantity of ecosystem services lost due to deforestation.

In order to quantify the ecosystem service lost, the volume of ecosystem services actually realized by humans (or potentially realizable in a realistic scenario), must be related to the area of forest lost each decade. Not all services are supplied by all forests. A further consideration is identifying the sustainable level at which services can be provided (i.e. for goods extracted from the ecosystem).

Models can be developed to estimate the quantity of ecosystem services associated with different land cover types and across spatial units. These may take into consideration factors such as the location of specific species, of human populations and infrastructure, geology and topography, soil type/quality, rainfall patterns (temporal and spatial), and information on extreme natural events. These models can vary in their complexity. In order to improve accuracy, additional data may need to be collected in the field. However, this is likely to be expensive, and unnecessary for broad order of magnitude estimates. The table below sets out the main elements required to identify where forest benefits will occur and then estimate the quantity.

Table 2: Estimating ecosystem service quantities (i.e. which are potentially realizable)

Forest ecosystem service	Restriction (identifying forest location)	Elements (estimating quantity)
Loss of NWFPs, specifically food, medicine, fibre and resin	Areas that can be accessed by a population. An average harvest distance buffer (e.g. 6 km) around human settlements can be defined (this may vary by NWFP).	Average harvest level for main NWFPs can be used, but ideally this should reflect a sustainable harvest level.
Loss of sustainably harvested wood	Areas that can be accessed by a population. An average harvest distance buffer around human settlements or roads can be defined.	A sustainable harvest volume of wood per land area unit needs to be identified.
Impact on pollination of crops bordering the forest	Cropping areas adjacent to forests (e.g. 1 km buffer), with crops benefiting from pollination.	Increased crop yields from wild pollination.
Regulation of diseases (to humans)	Forests in areas with human population, where diseases have been identified that increase in prevalence following deforestation.	Number of people falling ill
Carbon	All forest	Carbon density, based on forest type and soil carbon. Note: net benefit is based on the carbon difference with the alternative land use.
Impact on sedimentation through soil erosion	Areas at high risk of erosion (medium to high slope and significant rainfall), and	Resulting impact on relevant structures, e.g. reduced

	where infrastructure (such as dams) exists downstream of the catchment.	annual output from hydroelectric dams.
Impact on water purification – clean water for drinking (and fisheries)	Watershed areas upstream of human populations that rely on river water for drinking (or fishing).	Population affected multiplied by water usage (or fish catch).
Impacts on water regulation	Watershed areas upstream of human populations that experience flooding or rely on river water for irrigation and drinking, or hydro-electric dams (avoid double-counting sedimentation impacts).	Infrastructure and population impacted by increased high-flow. Population, crop yield and electricity production impacted by low-flow.
Loss of biodiversity/landscapes as a tourism resource	Scenic areas or locations containing iconic species that are accessible to tourists (or that provide key ecological corridors to such sites).	Identify existing tourist visitor numbers to sites, scenic areas, or locations containing iconic species and extrapolate.
Cultural values to national and global population	Natural forests containing iconic species or high biodiversity.	Importance to global population is likely to be a function of prominence.

For the additional services the following approaches may be useful:

Impact on risk reduction from mangroves as sea defences	Locations on the coastal margin, with hinterland containing human settlements or infrastructure close to sea-level (e.g. up to 1 metre above high tide mark).	Frequency of storm events multiplied by population impacted, and reduction due to mangroves.
Regulation of pests (natural pest control – net benefit)	Cropping areas adjacent to forests (e.g. 0.5 km).	Pest regulation impacts on crop yields minus impacts from pests associated with forests.
Impacts on local climate (shelter/shade)	Cropping areas adjacent to forests (e.g. 0.5 km).	Micro-climate impacts on crop yields.
Genetic resource	Areas of forest with high biodiversity value.	Average genetic resource per area of high-biodiversity forest.

viii. Quantifying the monetary value of these lost services.

There is a large collection of literature, written over many years, which explores the value of forests (Table 3). Initially, this examined their provisioning service value, mainly for wood. Estimates centred on maximizing the profits from harvesting, which is calculated as the value of wood minus any costs (such as cutting, sawing, transporting under different forest management approaches), and this is still the main concern of commercial forest operators.

Valuation of forests as a recreational resource can be traced back 65 years, to when the economist Harold Hotelling used the Travel Cost Method. Around the same time, another economic valuation approach, the Contingent Valuation Method, was proposed; though it was not first used until 1963 (for the value of a wilderness area to hunters). Valuation studies of ecosystem services beyond recreation began not long after; the first of these related to tropical forests dates to 1969. The number of studies slowly increased throughout the 1970s-80s, and then more rapidly in the 1990s.

Following the 1992 Rio Earth Summit it was agreed that improved national accounts that include valuations of natural capital and ecosystem services were needed. Over the last ten years momentum for this has increased and a number of countries have been developing forest asset accounts, often working in partnership with the UN. Costa Rica pioneered the use of formal Payments for Ecosystem Services mechanisms in 1997 by establishing a country-wide program called *Pago por Servicios Ambientales*, which aimed to reverse the severe deforestation rates existing in the country at that time (Pagiola, 2008).

Table 3: Evolution of forest monetary valuation

Interest in forest valuation and year of first use	
Forest management and planning	c1620, and c1820
Recognition of recreational value	1949
Valuation of wilderness hunting	1963
Biodiversity and ecosystem services	(1948), 1969 for first tropical forest study
Government national economic accounts	1992
Payments for Ecosystem Services (PES) schemes implemented	1997

Table 4 provides a brief overview of available economic methods to estimate a monetary value of forest ecosystem services. They are grouped under four broad headings, related to the type of economic approach adopted: direct market valuation, revealed preference, stated preference, and benefit transfer (though the latter relies on valuations using one of the other three approaches). Stated preference techniques can be the only way of estimating the value of certain services, but cannot be used with the SEEA Experimental Accounting framework, which expects values to be derived from market transactions.

Table 4: Valuation methods

Direct Market Valuation		
Market prices	Based on market data (quantities, prices, costs), which is relatively easy to obtain, and may already be available in previous reports. Can normally only be applied to benefits that have a direct link to a market.	(i) Wood and NWFP benefits can be valued through market prices, prices of substitute products, or the labour costs of collection. (ii) Carbon benefits can be valued through prices in current markets, or future carbon payments. (iii) Tourism benefits can be estimated using revenues from forest-based tourism; e.g., entrance fees, tour guide costs, accommodation.
Cost-based	Estimates the costs of negative impacts in a business-as-usual scenario as a way of identifying the value of environmental benefits.	(i) Lost agricultural production from increased incidence of droughts or floods in a deforestation scenario, or increased disease (e.g. malaria). (ii) Costs of switching to alternative water sources (or additional water treatment). (iii) Costs of dredging to regain water flow.

		(iv) Carbon in terms of estimated damages from global warming.
Production-based	Estimates the value of environmental benefits that can serve as an input in the production of a marketed good via production/profit functions or productivity changes.	(i) Pollination benefits can be gauged by valuing their contribution to crop production/revenues. (ii) Soil retention benefits (i.e. reduced erosion) can be valued through their contribution to hydropower generation via sedimentation control. (iii) Genetic resources can be valued as an input in the development of pharmaceutical products.
Revealed Preference		
Travel cost	Seeks to indirectly reveal people's willingness-to-pay for environmental benefits.	Calculates the value of environmental benefits from the time and travel costs that people incur to visit a place (in this case a forest area).
Hedonic pricing	Derives the implicit price for forest benefits by modelling observable prices using a set of explanatory variables relevant to the benefit.	Existence benefits can be valued by estimating consumers' willingness to pay price premiums for products, such as wood products with sustainability certification.
Stated Preference		
Contingent valuation	Based on sample surveys asking people how much they would be willing to pay for environmental benefits (or accept in compensation or their loss).	(i) Hydrological benefits can be valued through estimating a forest watershed household's willingness to pay for improved water supply. (ii) Recreational benefits can be valued through estimating the willingness to pay of tourists to visit forest with high biodiversity. (iii) Existence value can be gauged by asking people how much they are willing to pay to protect an area of forest.
Choice experiments	Ask a sample of people to choose between several hypothetical options. Each option is related to a number of attributes whose levels vary (one is monetary value, e.g. a fee or payment).	(i) Hydrological benefits can be measured by estimating farmers' value to them of sufficient water supply for agricultural irrigation during the dry season. (ii) The relative value of forest bird and wildlife species to tourists can be estimated. (iii) Value placed on the existence of unique forest species or landscapes can be estimated.
Benefit Transfer		
Benefit transfers	Uses existing valuation studies and transfers the value to the new site, making some adjustment. This reduces the need for expensive new surveys. The type of service valued will depend on the original study[14].	(i) 'Unit benefit transfer' takes the unit value of environmental benefits (e.g. per hectare) from original studies as a reference value. (ii) 'Adjusted unit transfer' adjusts the unit value by the characteristics of the new location (such as population income, or tree species in the forest). (iii) 'Value function transfer' applies parameters from an original study determining the importance of ecosystem characteristics in a value function.

[14] Based on the growing number of valuation studies, the Economics of Ecosystems and Biodiversity (TEEB) initiative has created a database of values to provide a starting point for applying benefit transfer methods (van der Ploeg et al., 2010).

An added dimension of the methodology developed in the present report is that historical valuations going back a number of decades are required. Based on the information above, Table 5 recommends methods for the valuation of the various priority forest ecosystem services.

Table 5: Recommended monetary valuation approach of forest ecosystem services

Forest ecosystem service	Monetary valuation approach
Harvested NWFPs, specifically: foods, medicines, fibres, and resins	Market prices: data collated from national reports, academic literature, FAO international commodity price data.
Sustainably harvested wood products	
Pollination of crops bordering the forest	Production-based: using data from reports and academic literature (note: may not be country or crop-specific).
Regulation of diseases (to humans)	Cost-based: using data on treatment costs from the literature (or cost of lost labour).
Carbon retention/sequestration.	Cost-based: use the Social Cost of Carbon to estimate damages (various SCC estimates are available, though not for historical periods).
Reduced sedimentation through limiting soil erosion	Production-based: data on location of dams, generation capacity, and electricity prices to estimate the impact on hydro-electric power production.
Impact on water purification: clean water for drinking and fisheries	Cost-based: additional drinking water purification costs, lost fisheries output, or stated preference where country studies exist.
Impacts on water flows: floods and droughts	Cost-based: impact of additional flood damage[15] on capital assets and output, as well as the impact of increased droughts on crop and livestock (plus sectors beyond farming).
Biodiversity and landscapes as a tourism resource	Market prices: tourist expenditure, or benefit transfer / existing stated preference studies when forest eco-tourism has yet to be developed in that area.
Existence and bequest values to national and global population	Existing contingent valuation, choice experiments, or hedonic pricing studies reported in the literature (note: unlikely to be available for the specific country).

In all cases, where data for estimating the value of a forest ecosystem service is unavailable, the findings from previous studies that have calculated costs or benefits in the country, or neighbouring countries, may be used. Whilst data for the last decade is likely to be relatively easy to obtain, for decades further back in time the data will become increasingly sparse. Back-casting will therefore be required. Since the value of many ecosystem services associated with a particular area of forest land will vary through time due to changing population levels, land-use patterns and commodity prices, to

[15] As a direct result of deforestation –distinct from the result of urban expansion on to flood plains.

fully value the ecosystem services over the past 50 years an alternative (counter-factual) development scenario would need to be modelled.

An easier way forward is to divide the ecosystem services into three types: 'carbon', 'access-independent', and 'access-dependent' services. Carbon retention is considered separately as a service with a once-only (or 'pulse') occurring impact, which is costed as the carbon emissions resulting from deforestation and land conversion, at that particular point in time (with the impacts of the emissions valued up to some future point in time, e.g. end of the 21st century (see Appendix A for further detail)). The access-independent forest ecosystem services would have existed if the forest had been retained given the observed historical development path. So for instance, reducing soil erosion, the various hydrological benefits, as well as the existence and bequest values would all have existed for an area of forest until now and on to the end of the 21st century[16]. Access-dependent services, on the other hand, depend on the pattern of deforestation. This is because some services would have been unused, e.g. NWFPs are only readily accessible in areas close to villages or roads, and pollination benefits only occur where forests buffer croplands. Thus the analysis needs to consider the spatial pattern of access for these services over time as deforestation moves these boundaries.

ix. Estimate the corrected economic gains from deforestation.

In order to calculate net benefit, the costs and benefits must be compared. In this instance the benefits are the wood products and agricultural production resulting from deforestation, and the costs are the lost ecosystem service benefits associated with the lost forest area. It may be useful to present the information in graph form. A sensitivity analysis to test the importance of the various assumptions used on the results will be required.

x. Compare the results of step ix with the information gathered in the first step.

Once the analysis in ix is completed it is then useful to explore whether there were additional benefits from development in terms of social improvements that can be clearly linked with the land-use change, as well as analysis of the pattern of development. This is in order to determine whether economic growth fuelled by land-use change has benefited the population as a whole or whether the benefits have been captured by a small minority at the expense of the wider population. When statistical information is unavailable, a qualitative analysis would then be required.

[16] So long as there was a hydro-electric dam sited downstream in the case of soil erosion, and human population downstream in the case of hydrological benefits, and on-going global concern for tropical forest conservation for their existence value. Values could be modified for previous decades to take into account changes in population, or the year that dams were constructed, etc.

6. Case study: analysis of northern Argentina (1960-2010)

6.1 Geography and land use

The country chosen for detailed analysis is the Republic of Argentina[17]. Argentina is the second largest country in South America and eighth largest in the world. It has a wealth of natural resources and was first settled (on the southern tip of Patagonia) around 13,000 years ago (Gil et al., 2005). Spanish domination of this region began in 1516 until the declaration of independence was made (in 1816) and a federal state was formed in 1853-1861 (Bethell, 1984).

Located in the southern hemisphere, Argentina is a long country, stretching from the subtropics in the north to the sub-polar region in the south. The climate reflects this, from warm/tropical to temperate to cold, moving north to south; a large arid zone cuts across the country from west to east. It is bordered by Chile in the west, Bolivia and Paraguay in the north, and Brazil and Uruguay in the north-east. The western edge of the country runs along the Andean Mountains.

Argentina can be divided into five broad agricultural regions:

- Mountain: the Andes along the western border, with glacial mountains, lakes and arid basins, generally unsuitable for agriculture.
- Foothills: the sub-Andean irrigated enclaves suitable for growing fruit.
- Patagonian plateau: low, arid, pastoral steppes, which are suitable for extensive sheep ranching, with fruit and vegetable production in the valleys.
- Chaco plain: fertile lowland with subtropical rainforests, being converted to farmland.
- Central Pampas: fertile plains (both humid and semi-arid areas) which provide much of Argentina's agricultural land and output, including cattle, wheat, corn, and soya.

The major remaining forests at risk are situated in the northern to central parts of the country; therefore the following analysis will be focused on this area (hereafter referred to as 'northern Argentina', see Figure 1, below).

Argentina's population has been expanding (Table 6), to around 41 million at present. Over 90% of live in urban areas, with the majority located in northern Argentina (see Figure 1).

Table 6: Argentina's population 1970-2010

Year	1970	1980	1990	2000	2010
Total population (millions)	24.0	28.1	32.6	36.9	40.4

Source: UN (2010).

[17] Please note that this is a desk study only, and that the analysis was carried out before the launch of Argentina's UN-REDD National Programme in July 2015

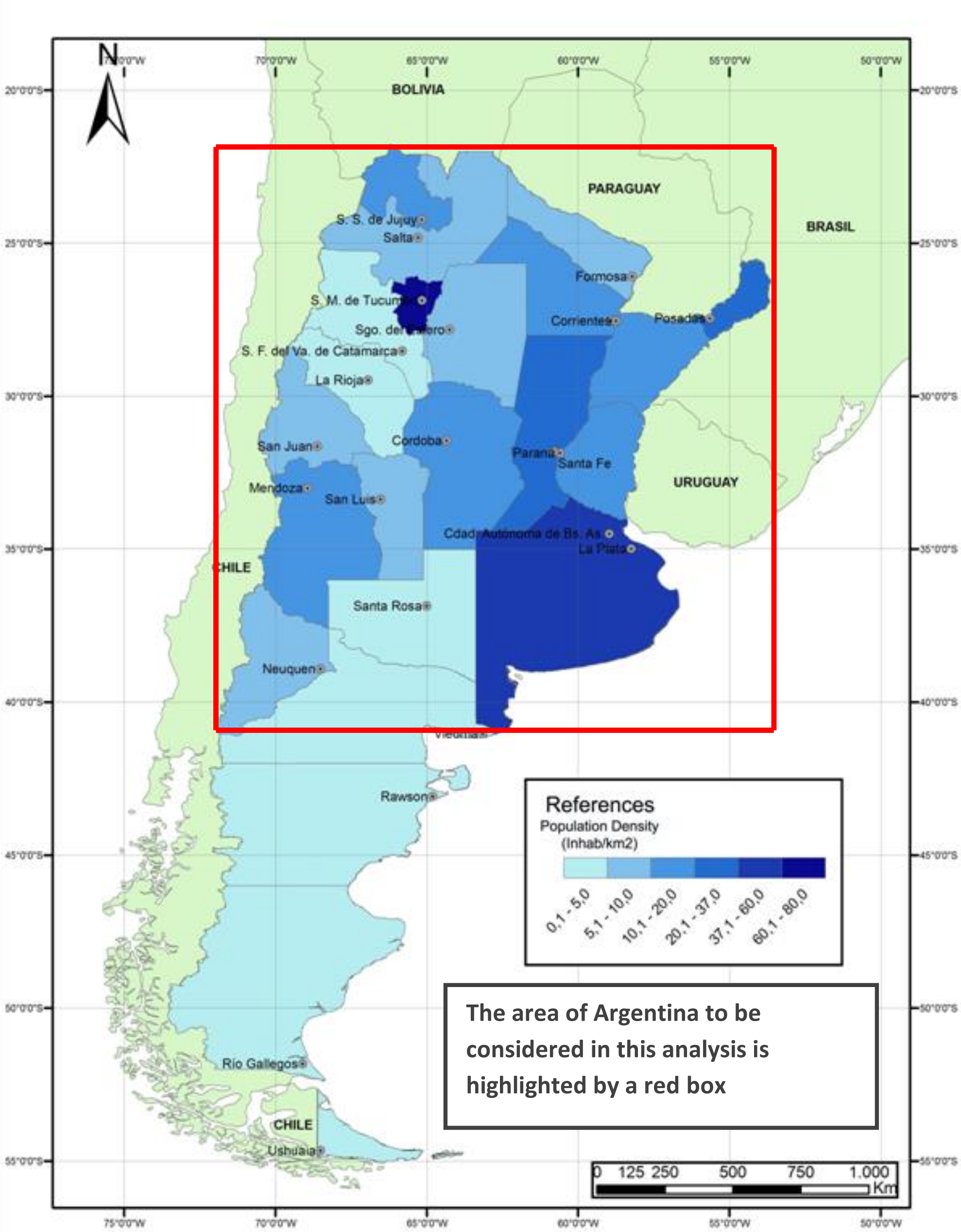

Figure 1: Population Density (2010 Census), highlighting the study area for this report

Source: INDEC (2013).

6.2 The economic situation in Argentina and the role of forests

Argentina has a diversified industrial base[18] and a highly literate population, but large external debts and episodes of difficult economic management have led to periodic economic crises. With two government debt defaults since 2000, the resulting weaker currency has been a driver of inflation. The number of people below the poverty line of US$2 a day peaked in 2002 (World Bank, 2014a), at close

[18] Including food processing, motor vehicle and consumer durables manufacturing, textiles, and chemicals.

to a quarter of the population. The Gini index then indicated that Argentina was one of the most unequal countries in the world. Since then poverty has decreased so much that the Gini index shows Argentina to have become one of the most equal societies in Latin America. GDP per capita was US$14,760 in 2013 (World Bank, 2014a).

The main driver of deforestation is land-use change for agriculture. Argentina is the world's fifth largest exporter of agricultural products, and this accounts for around 9% of GDP (Europa World, 2014). Agricultural mechanization is common amongst large-scale farming enterprises, but many farmers still use minimal chemical inputs and livestock are usually kept extensively. As a result, Argentina is the world's third largest organic producer of cereals, fruits, vegetables, sugar and wines, which are mainly exported. It is also the world's number three soya bean exporter (Reuters, 2014). In contrast, wood product exports provide a small source of income for Argentina, with a 0.5% share in the national GDP (Global Forest Watch, 2014). According to FAO, around 11 million cubic metres of industrial roundwood were harvested in 2010, mainly from plantations (FAO, no date).

Although not directly relying to a significant extent on forests[19], Argentina's economy (as with most countries) has indirect linkages to forests, such as through the hydrological functions which forests can provide. One clear example is reducing downstream siltation into hydro-electric dams. Around a quarter of Argentina's electricity is supplied by hydro-power (World Bank, 2014a) (see Table 7); this is considered in further detail in section 6.6.

Table 7: Hydro-electric capacity and output in Argentina 1970-2010

Year	1970	1980	1990	2000	2010
Capacity of hydro-electric power station (GWe)	0.61	3.63	6.61	9.60	10.05
Hydro electric output (GWh)	1.5	15.1	17.9	28.8	33.6

Source: World Bank (2014a)

6.3 Forest cover and drivers of land-use change

A census in 1914 found that Argentina had approximately 105 million hectares of forest (IPCC 1997). Currently, about 27 million hectares is forested according to the UN Food and Agriculture Organization (FAO, 2015), or 39 million hectares according to Global Forest Watch (Global Forest Watch, 2014). According to FAO, some 16.2 million hectares of Argentina's forest cover were lost between 1980 and 2000. Between 1990 and 2015, Argentina lost an average of 307,200 hectares of forest per annum (FAO, 2015), which is equivalent to an average annual deforestation rate of around 1% (FAO, 2015), although others record a higher loss (Global Forest Watch, 2014). Over this period, rates were highest between 2000 and 2010.

[19] Around 70,000 people were employed in the forest sector in 2011 according to FAO data (Global Forest Watch, 2014), of 10 million people being employed in total (Europa World, 2014) (i.e. 0.7%).

Most of Argentina's forests are regenerated (secondary) forest, with a small percentage of primary forests (Butler, 2006). This reflects the presence of human populations modifying the landscape for several thousand years. Following European settlement of the dry forest area in the 1880s-90s for beef production, many trees were cut down and used for timber (including sleepers for railways and fence-posts for ranches) or firewood. For example, of the 10 million hectares of forest found in Córdoba a century ago, only 12% remain (Valente, 2005). A difference from the situation in many countries is that the greatest deforestation occurred in the hills and mountains in this region, where only 2% of the native forest cover survives (Valente, 2005).

Forests are mainly being lost to agriculture. Deforestation is centred in the north of the country where the greatest area of forest (the continent's largest after the Amazon), is located. The importance of reducing deforestation has been recognized by the government of Argentina. A sustainable forest management strategy is currently being devised, funded by a loan from the World Bank which runs from March 2009 to September 2015 (World Bank, 2008). Argentina's UN-REDD National Programme was launched in July 2014 to protect native forests by reducing emissions caused from deforestation and forest degradation and to promote climate change mitigation (UN-REDD Programme, 2015). However, forest loss continues, with deforestation in the Chaco region receiving recent publicity.

Proba-V image from 4 February 2014 showing patterns of deforestation within the region of Chaco, Argentina. 25° 5'29.79"S 62°34'6.58"O. http://www.esa.int/spaceinimages/Images/2014/05/Deforestation_in_Argentina

Deforestation has been particularly severe in the seasonally dry Chaco forest, and the period 1969-1999 saw one of the highest rates in recorded history for an area of forest of its size, with 85% of the total area cleared (Zak et al., 2004). Another important area is the Interior Atlantic Forest, in Misiones province, which covers only one percent of Argentina's total land area, but is the country's most diverse ecoregion, holding 3,148 taxa of vascular plants and 1,124 of vertebrates, including 52 taxa of vascular plants and 9 of vertebrates exclusive to Argentina (Pavedano et al., 2003). Hence, the Interior Atlantic Forest is an area of megadiversity for the world as a whole.

Between 2004 and 2011, over a million hectares of forest were converted to agricultural land in just two provinces, Santiago del Estero and Salta (Pollock, 2012). The main driver is production of soya, a very profitable crop. Soya production in Argentina has increased rapidly, with exports increasing ten-fold between 1977 and 2000 (Ray, 2001). In the 2013/14 crop year, 53 million tonnes of soya bean were harvested, an increase of 9.5% over the previous year (Reuters, 2014). Although GM soya can be grown intensively (i.e. all the year round), producing high yields per hectare, it is becoming a significant driver of deforestation as the area of land under cultivation increases (Leguizamon, 2014). Large-scale agribusinesses are encroaching on the forest directly, and also by displacing cattle ranchers into forested lands (Carey and Oettli, 2006). Most soya production has occurred in the central north area (see Figure 2, below).

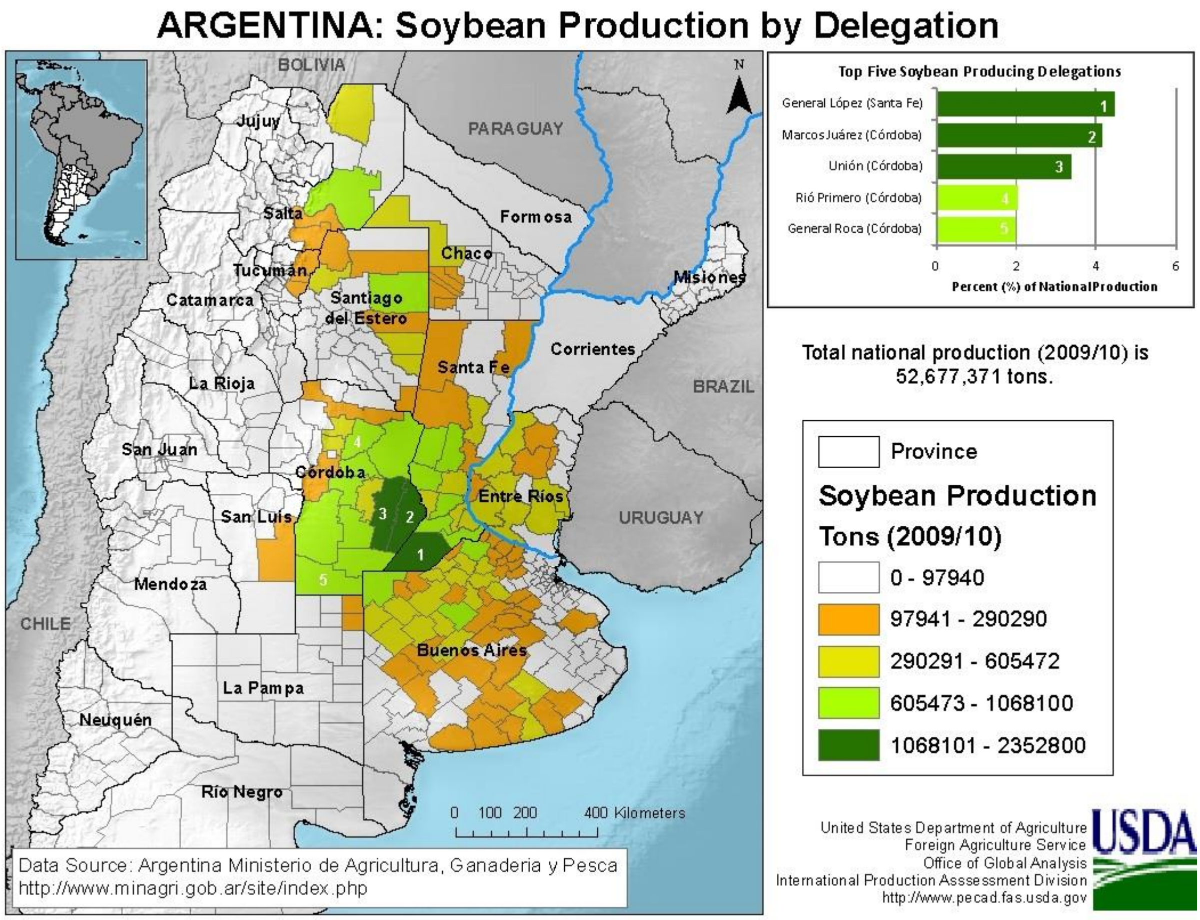

Figure 2: Argentine soya production by region

Source: USDA (2011).

6.4 Estimation of wood products and agricultural output and value as a result of deforestation

The area of forest land in northern Argentina that has been converted into different uses was estimated using the HYDE data set (Table 8, see section 6.6 for more detail). Soya and beef production were identified as the two main drivers of recent deforestation, with Argentine agriculture now being dominated by mechanised production of soya. Since beef and soya have been very profitable agricultural land uses (demonstrated by the shift from corn to soy bean cultivation), the choice of these for our valuation of the historical benefit of deforestation may over-estimate the value of some historical agricultural uses. A more detailed analysis that assigns land to all the different crops being grown is possible, but would be time-consuming.

Table 8: Estimated gains in land use from deforestation (ha) for each decade
(NB: 250,000 of cropland established in the 1960s was abandoned in the 1970s))

	Cropland	Pasture	Grassland
1960 - 1970	5,000,000	750,000	0
1970 - 1980	250,000	250,000	250,000
1980 - 1990	3,250,000	2,000,000	250,000
1990 - 2000	2,500,000	750,000	250,000
2000 - 2010	1,500,000	1,000,000	0

These estimates were compared with others to check whether they were realistic. Grieg-Gran et al. (2007) estimate that 2.3 million hectares of Chaco vegetation were cleared for soy cultivation between 1995 and 2005; they also estimate that just over one million hectares of additional grazing land became available for use between 1994 and 2003. Much of the 1990-2000 conversion to cropland shown in Table 8 could have occurred towards the end of the decade as soy planting rapidly expanded, so all these figures appear comparable.

The average soya yield for the most recent decade was projected back for previous decades (see Table 9) based on the trend in crop yield increases over the last 50 years - i.e., a doubling. Whilst past trends suggest that yields could increase in future, climate change and increasing soil erosion may counteract this. Degradation of soils due to soy cultivation is already occurring (Bronstein, 2013). Therefore, a constant yield is used from 2011 to 2099.

Table 9: Average soya yields (tonnes per hectare) for each decade (estimate)

	Average yield (tonnes/ha)
1960 - 1970	1.15
1970 - 1980	1.44
1980 - 1990	1.73
1990 - 2000	2.01
2000 - 2010	2.30

The likely stocking density for beef cattle on the new pastureland was assumed to be the carrying capacity, 0.7 cattle per ha on unimproved natural pastures and 2 cattle per ha on improved pastures (Grieg-Gran et al., 2007). The annual offtake (the percentage of herd slaughtered each year) is 15% on average (Garbulsky and Deregibus, 2006), and average carcass weight is 220 kg (Joseph, 2011). This carcass weight is used for historic and future years.

The estimated additional cattle numbers that could be farmed on the new pastureland were compared with an estimate from Grieg-Gran et al. (2007), who found, using official figures for each province, that the cattle population in the northern cattle expansion zones increased by about 1,500,000 head between 1994 and 2003. Our analysis estimated that there were an additional 1,675,000 cattle between 1990 and 2000. These estimates from overlapping time periods are fairly consistent with one another.

With regard to wood products harvested when forests are converted to agricultural land, the tropical forests of South America have harvests of only 5 to 30 cubic metres per hectare (Contreras-Hermosilla, 1999). These forests have a higher stem density but lower species diversity and smaller tree diameters than the moist tropical forests of Africa and Asia, resulting in a lower volume of commercial species. Wood processing results in further losses, so a usable yield of 80% of these values is adopted for our analysis in both shrub forest and tropical forest (Table 10). This is a yield at the optimistic end of the spectrum.

Table 10: Useable wood harvested as a result of land-use change each decade (estimate)

	m3 timber
1960 - 1970	23,000,000
1970 - 1980	8,000,000
1980 - 1990	27,000,000
1990 - 2000	19,000,000
2000 - 2010	30,000,000

The value of all these outputs is calculated by multiplying volumes by unit prices (which assumes that all agricultural inputs originate in Argentina). Using national and international data, the prices for soya beans and beef in Argentina for each decade were estimated as follows in Table 11:

Table 11: Prices (decadal mean) for soya beans (left), and beef (right), in Argentina (US$ per tonne)

	US$ / tonne (2010 prices)		**US$ / tonne** (2010 prices)
1960 - 1970	119.08	1960 - 1970	754
1970 - 1980	126.27	1970 - 1980	1,455
1980 - 1990	133.91	1980 - 1990	2,040
1990 - 2000	142.00	1990 - 2000	1,888
2000 - 2010	150.00	2000 - 2010	1,966

Estimated future crop prices were explored as this data is required up to the year 2099, but for obvious reasons few studies have tried to make price estimates for such a long period into the future. Up to 2050, soy prices are expected to plateau in real terms at current levels (Kruse, 2010), so for our study it is assumed that prices hold steady until 2099. Since demand for meat increases with growing affluence, it is assumed that beef prices are likely to increase. Estimates for future growth in demand for beef are only up to half of historical growth (Alexandratos and Bruinsma, 2012), so it is assumed that beef prices in Argentina increase by half their historical rate (i.e. 250% up to 2099). The cumulative present values were then calculated in decadal increments.

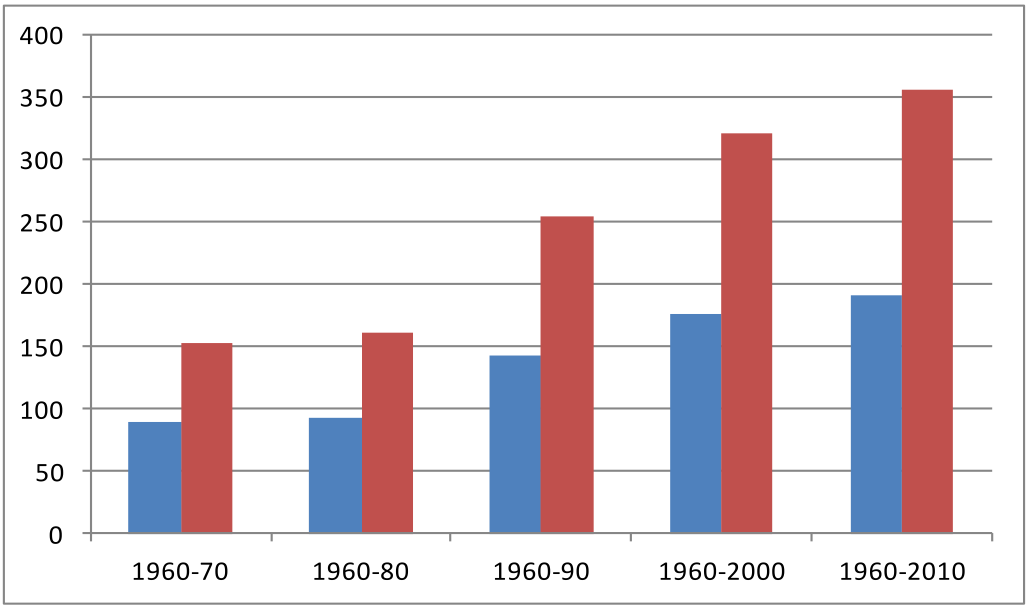

Figure 3: Cumulative value (up to 2099) of land converted from forest and used for crop production – with soy as a representative arable crop (2010 US$ billion), red bar 1% discount rate, blue bar 5% discount rate

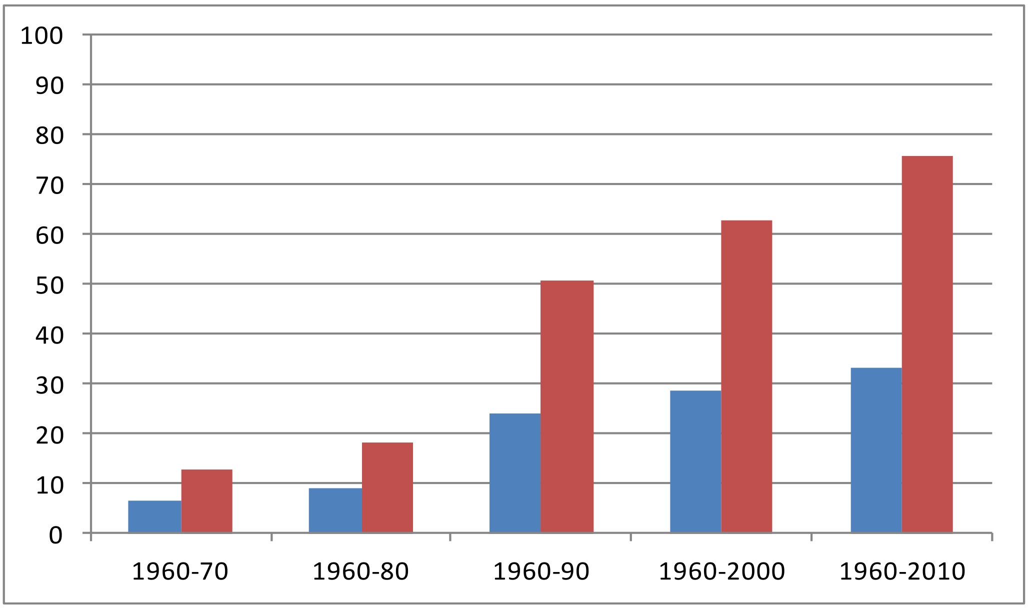

Figure 4: Cumulative value (up to 2099) of land converted from forest and used for pasture and grassland – with beef as a representative livestock product (2010 US$ billion), red bar 1% discount rate, blue bar 5% discount rate

Prices for timber going back to 1960, in year 2010 values, were obtained from the World Bank Commodity Price Data (The Pink Sheet) (see section 5), and these were adjusted based on the price difference between the World Bank series and the timber price in Argentina in 2005 (Cubbage et al., 2007). However, little of the wood harvested from native forests is exported, and much of it, being of relatively low quality, may be used for on-farm purposes such as fencing or building. Thus a value of 50% of the international trade price is assumed[20]. Unlike the other harvest values examined, the wood

[20] Given the values in Figure 5, using the international trade price would make an insignificant difference to the overall value of the land-use change.

harvested during land clearance happens only once, hence the values relate only to the period in which they occur. The calculated values are presented in Figure 5 below[21]:

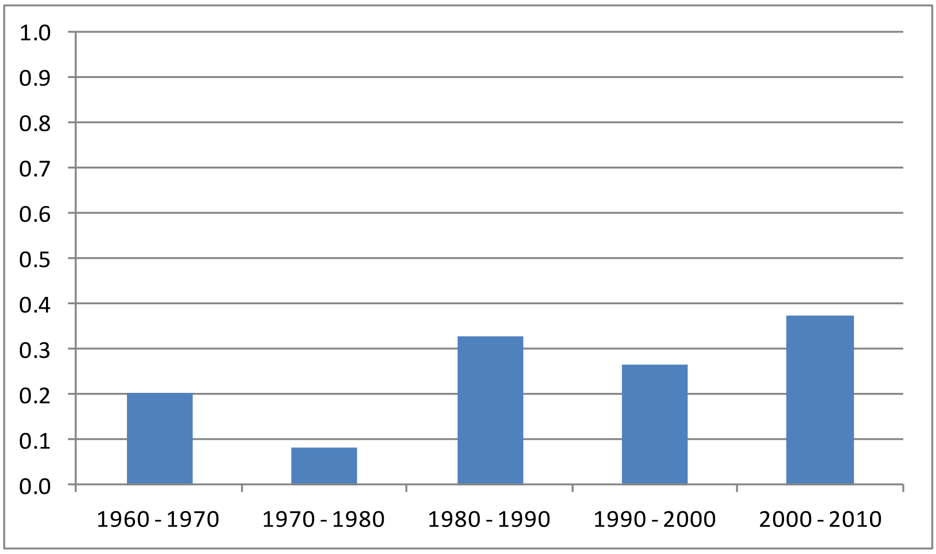

Figure 5: Value of harvested wood for each decade (US$ billion) (estimate)

6.5 Identification of ecosystem services related to forests in Argentina

Following an exploration of the role of forests in Argentina, the ecosystem services for analysis have been identified as:

1. Harvested NWFPs,
2. Sustainably harvested wood,
3. Pollination of crops bordering the forest,
4. Regulation of diseases,
5. Carbon retention/sequestration,
6. Reduced sedimentation through limiting soil erosion,
7. Impacts on water flows: floods and droughts,
8. Biodiversity and landscapes as a tourism resource.

These are outlined in more detail below.

[21] If a (possibly more realistic) lower wood utilization rate of 40% is used, then the value of harvested wood over the entire period (1960-2010) is reduced to just US$0.6 billion.

Harvested NWFPs

Data on NWFPs in 2012 is available from the Environment Ministry, both for the amount produced (in tonnes), with a detailed description for each product and per region, and the price of the products (SADS, 2012). Significant amounts of resin have been harvested in recent years (2006-2012) (SADS, 2014). A number of studies of local use exist. For example, the Valdivian temperate rainforest, mostly found in Chile, extends into Argentina. Tácon *et al*. (2006) note various NWFPs being used here, with edible fungi being particularly valuable. Yerba mate (*Ilex paraguariensis*) was originally collected from its natural habitat here, but has since 1903 been cultivated for consumption. In another example, Suárez (2010) identifies NWFP use among the Wichí people of the Chaco Salta province as follows:

Table 12: NWFP use by Wichí people in Salta province

NWFP* * most NWFPs are used in medicine, food and fodder	**%**
Trees (includes arboreal cactus)	34
Herbs	22
Shrubs	21
Vines	8
Non-arboreal cactus	5
Epiphytes	5
Fungi (includes lichens)	3
Lianas	2

Source: Suárez (2010).

The estimated total production and value of NWFPs in Argentina in 2011 can be seen in Table 13:

Table 13: Production and value of NWFPs in Argentina (2011)

Use	Production (in tonnes)	Value (US$)
Industry	16 730	107 222 570
Food	1 300	11 601 052
Craft/ ornamental	568	704 540
Construction	299	92 446
Fodder	299	415 000
Seeds for forestation	230	175 935
Textiles	10	22 000
Medicine	10	11 008

Source: SADS (2014).

Sustainably harvested wood

The value of primary and processed forest products for 2011 is given in Table 14. Forests provide 10% of the raw material for industry, with plantations (primarily pine, eucalyptus and willow) providing 90% of this amount, despite covering only 1.2 million hectares (Ministerio de Ciencia, Tecnología e Innovación Productiva, 2013).

Table 14: Value of primary and processed forest products in Argentina for 2011

	Extraction (tonnes)	Value (US$)
Primary products	2 952 487	698 952 872
Logs	465 325	149 660 831
Fire-wood	2 474 698	530 300 610
Stakes	12 464	18 991 431

Source: SADS (2014).

These data do not show whether or not the harvest from primary forests is sustainable. Sustainable forest management has been promoted in recent years, for example through a joint initiative between the Canadian Forest Service and the Argentine Model Forest Programme and a World Bank project with a focus on conservation and small producers (World Bank, 2014c). A US$60 million loan from the Inter-American Development Bank aims to foster sustainable plantations, increase product quality, diversify the productive base, promote forest certification and improve the access of micro-, small- and medium-size enterprises to supply chains and markets (IDB, 2012).

Pollination of crops bordering the forest

Around 40% of global food production comes from the 87 of 115 leading global crops that depend on insect pollination (Klein et al., 2007). It is often the more valuable crops by weight or volume that rely upon or benefit from insect pollination. A number of studies have attempted to quantify the value of pollination services to agriculture. At the global level, estimates cluster around 10% of the value of agricultural production (Stathers, 2014); however, this will vary by country depending on the crops grown. Whilst there is limited information available specifically related to Argentina, the value of pollination from forest species has been recognized in other South American countries. Ricketts et al. (2004) explore the economic value of tropical forest for coffee production by using pollination experiments along distance gradients. They found that forest-based pollinators increased coffee yields by 20% within ≈1 km of forest, and also improved coffee quality near forest edges, by reducing the frequency of small misshapen seeds by 27% (Ricketts et al., 2004). In Brazil, Freitas et al. (2014) posited that pollination deficit could cause low yields in cashew (*Anacardium occidentale*) with deforestation surrounding cashew plantations preventing pollinator visits. Cashew nut yield was highest when plantations bordered a small forest fragment and were close to the large forest (Table 15).

Table 15: Pollinator visits and subsequent crop yields for cashew plantations close to (green) and far from (pink) a large forest area (for cashew plantations not bordered by small forest fragments)

Distance to nearest large forest (m)	Number of native pollinator visitors	Average crop yield (g)
432	102	3420
465	55	1808
2513	39	780
2569	38	819

Source: Freitas et al. (2014)

The degree of reliance upon insect pollination of the principal crops grown in Argentina are listed below (Table 16).

Table 16: Crops grown in Argentina, production and pollinator reliance

Crop	Production ('000 tonnes)	Pollinator reliance
Sugar cane	26 960	No
Maize	23 800	No
Wheat	14 501	No
Sorghum	4 458	No
Barley	4 086	No
Grapes	2 890	No
Potatoes	2 127	No
Rice	1 748	No
Oranges	877	No
Tomatoes	699	No
Oats	415	No
Sweet potatoes	390	No
Carrots & turnips	244	No
Bananas	172	No
Soybeans (Soya beans)	48 879	Some benefit
Groundnuts	702	Some benefit
Tangerines, satsumas, etc	401	Some benefit
Beans	333	Some benefit
Chillies & peppers	133	Some benefit
Sunflower seeds	3 672	Yes
Lemons and limes	1 229	Yes
Apples	1 116	Yes
Pears	691	Yes
Pumpkins, squash & gourds	338	Yes
Peaches & nectarines	285	Yes
Grapefruit	189	Yes
Olives	170	Yes
Plums and sloes	148	Yes
Watermelons	124	Yes
Cantaloupes & other melons	77	Yes

Source: pollinator assessment synthesized from: http://bees.techno-science.ca/english/bees/pollination/food-depends-on-bees.php, http://plants.usda.gov/pollinators/Native_Pollinators.pdf and http://www.foxnews.com/leisure/2012/07/19/crops-that-would-disappear-without-bees/. Production data from FAO statistics website: http://faostat.fao.org/DesktopDefault.aspx?PageID=339&lang=en&country=9

Regulation of diseases

It has been recognized for some time that deforestation can increase disease risk. Tree removal and new logging roads leave behind pools of standing water, and increased sunlight (allowed by the destruction of the forest canopy) warms the pools of water, leading to growth of algae as well as reduced water acidity, creating an ideal growing environment for mosquitoes, thus increasing malaria and dengue rates (Waite, 2008). Dengue fever infects about 400 million people worldwide each year, and is one of the primary causes of illness and death in the tropics and subtropics, as well as incurring a high cost burden (about US$50.4 million in Mesoamerica alone) (Proyecto Mesoamerica, no date). Over 26,000 people fell ill with dengue in 2009 in Argentina, though the wide annual fluctuation is reflected in the fact that just 900 cases of dengue were reported in 2010 (World Bank, 2014d). Table 17 shows diseases that may spread in Argentina as a result of deforestation.

Table 17: Examples of deforestation-associated infectious diseases relevant to Argentina

Agent/disease	Distribution	Hosts	Exposure	Possible emergence mechanisms
Viruses				
Yellow fever	Africa South America	Non-human primates	Vector	Deforestation and expansion of settlements along forest edges
Dengue	Pantropical	Non-human primates	Vector	Deforestation and expansion of settlements
Simian immunodeficiency virus (SIV)	Pantropical	Non-human primates	Direct	Deforestation and human expansion into forest Hunting and butchering of forest wildlife
Rabies	Worldwide	Canines Bats Other wildlife	Direct	Human expansion into forest
Protozoa				
Malaria	Africa Southeast Asia South America	Non-human primates	Vector	Deforestation, habitat alteration beneficial for mosquito breeding Human expansion into forest
American cutaneous leishmaniasis (ACL)	South America	Numerous mammals	Vector	Deforestation and human expansion into forest Habitat alteration, habitation building near forest edge
Bacteria				
Lyme disease	Worldwide	Humans Deer Mice	Vector	Possible association with deforestation and habitat fragmentation
Leptospirosis	Worldwide	Rodents	Indirect	Watershed alteration, resulting flooding

Source: Wilcox and Ellis (2006).

Emerging infectious diseases are considered to be among the biggest challenges to science and human development. The role of forest management is significant, as deforestation can lead to increased prevalence of zoonotic (i.e. originating in animals) and vector-borne diseases, as well as multiplying the prevalence of diseases capable of producing catastrophic pandemics (Cho, 2014). According to the United States Agency for International Development, nearly 75% of all new emerging diseases affecting humans are zoonotic (e.g. AIDS, SARS, H5N1 avian flu, and H1N1 flu). Increasingly, wild animals which may have carried diseases without effect for years are coming into contact with humans, often because of deforestation (Cho, 2014).

There is evidence of an increased disease burden in Argentina as a result of historic deforestation. American cutaneous leishmaniasis (ACL) has significantly increased its incidence in Argentina during the 1980s (Salomón et al., 2012). ACL is endemic in four bioecological tropical/subtropical regions (i.e. the foothills of Yungas Forest, Dry Chaco, Wet Chaco, and Paranaense Forest), and the cases of ACL were usually related to a 'common source' like deforestation (Salomón et al., 2012). The reports by year usually fluctuate between 40 and 90 for the whole country, but since 1985 the cases clustered in outbreaks of up to 900 cases, beginning in the northwest (Yungas) and reaching the eastern border of the endemic area by 1998 (Salomón et al., 2012). The last epidemic year was 2002, with 748 human cases throughout the endemic provinces. Whereas previously forest was logged and abandoned in the foothills of Yungas and central Chaco, settlement for agriculture creates additional risks. A 'border effect' from deforestation and subsequent construction of farm buildings exists, which is associated with greater abundances of vectors (Salomón et al., 2012). This change also pushes the adaptation of the epidemic vector of ACL, *Nyssomyia neivai,* to modified human landscapes, thereafter becoming prevalent and abundant in rural and peri-urban settlements (Salomón et al., 2012). Each disease case can bring a high economic loss, including the cost of the drug treatment, health staff costs and the temporary loss of income (and well-being) of patients.

Carbon retention/sequestration

Carbon plays a key role in driving climate change. Terrestrial ecosystems can add to or remove carbon from the atmosphere. They can accumulate carbon in both biomass and the soil. Forests can contain very high stocks of carbon per unit area, which are greatly reduced as a result of land conversion to cropland. The amount of carbon stored in different types of forests varies, and there will often be large in-country variations, as can be seen in Table 18 for four forested regions in Argentina.

Table 18: Carbon content (average t/ha) for four forested regions in Argentina: Chaqueño Park (PCH), Selva Misionera (SM), Selva Tucumano Boliviano (STB) and Bosque Andino Patagónico (BAP)

Carbon content (t/ha)	PCH	SM	STB	BAP
Above-ground	49.40	133.12	86.44	270.10
Below-ground	13.34	31.95	20.74	64.82
In deadwood	6.92	14.64	9.51	37.81
In leaves	2.80	2.80	2.80	16.00
In soil	38.00	47.00	65.00	95.00
TOTAL	**110.45**	**229.51**	**184.49**	**483.74**

Source: Gasparri and Manghi (2004).

Gasparri et al. (2008) explored carbon loss in northern Argentine forests between 1900 and 2005. They found that over that time almost 30% of the forests (10.6 million hectares) were destroyed, releasing

945 million tonnes of carbon. The amount of carbon (as CO_2) released partly depends upon the new land use. When forests are cleared for conversion to agriculture most of the above-ground biomass is usually burned, releasing the majority of carbon rapidly into the atmosphere. Some of the trees may be used as wood products, and as such these carbon stocks could thereby be preserved for a longer time depending upon the use. Forest clearing for agriculture also accelerates the decay of dead wood and litter, as well as below-ground organic carbon (IPCC, 2000). Whilst local climate and soil conditions will determine the rates of decay, in tropical moist regions most of the remaining biomass decomposes in less than 10 years (IPCC, 2000). For simplicity, it is often assumed that the post-deforestation carbon stocks in above-ground biomass pools are zero for land-use classes such as annual croplands (and associated roads and farm settlements), which represents a Tier I approach under the IPCC (a different approach is adopted in the current case study).

With regard to soils, the effect of land-use changes on soil organic carbon is poorly quantified due to insufficient data quality (only concentrations and not stocks, with shallow sampling depth, and limited representativeness) (Don et al., 2011). In a meta-analysis of 385 studies on impacts of land-use change in the tropics Don et al. (2011) found that the highest soil organic carbon losses were caused by conversion of primary forest into cropland (−25%) and perennial crops (−30%), but forest conversion into grassland also reduced soil organic carbon stocks by 12%. Secondary forests stored less soil organic carbon than primary forests (−9%) (Don et al., 2011).

Although certain arable farming methods associated with GM cropping (including soya in Argentina), such as conservation tillage (also known as 'no-till'), have been promoted as improving soil carbon balances, the evidence is mixed. The evidence in favour is based on experiments where changes in carbon storage have been estimated through soil sampling during tillage trials, to a depth of 30 cm or less (Baker et al., 2007). But where sampling extended below 30 cm, conservation tillage has shown no consistent accrual of soil organic carbon, instead showing a difference in the distribution (with higher concentrations in deeper layers under conventional tillage), and long-term continuous gas exchange measurements have also been unable to detect any carbon gain due to reduced tillage (Baker et al., 2007). Therefore, it is likely that arable, including soya, conversion of forest in Argentina is likely to lead to soil organic carbon losses of the order found in the meta-analysis by Don et al. (2011).

Reduced sedimentation through limiting soil erosion

Forests can help stabilize soils and thereby reduce erosion rates that lead to river sedimentation, which can be especially important for navigation or when hydro-electric dams are located downstream (there are also links to flooding, see below). Without tree roots to anchor the soil, and with increased exposure to sun, the soil can dry out, making it susceptible to both wind and water erosion. Research on historical erosion in Central America has found that the biggest rates of erosion occur after the initial forest clearance (Anselmetti et al., 2007). Numbers from FAO research (related to erosion caused by various factors, not just deforestation) (FAO, 1993) indicate that in Argentina erosion was affecting approximately 25 million hectares in 1993. The area affected by erosion had increased by 223,000 ha per year since the 1950s. Deforestation is a major cause of desertification in arid and semi-arid areas (Huss, no date)[22] and elsewhere it has been found that deforestation and over-cutting of vegetation are the predominant causes of soil degradation by both wind erosion and water erosion (FAO, 1994), thus it seems likely that much soil erosion occurring in Argentina is due to deforestation. Soil erosion can have large economic impacts, for example the erosion taking place at the Ventana hydrographic basin (south west Buenos Aires) results in agricultural output losses of US$217 million per year (Gasparri et al., 2008). However, the land is only available to agriculture because it was deforested, and so the in situ agricultural impacts of erosion are not estimated in this study.

Impacts of forests on water flows: floods and droughts

The ways in which deforestation is thought to increase flood risk include:

i. *sediment release* - recently cleared forest land is at greater risk of soil erosion (see above), especially during high-precipitation events, which can silt rivers and thus raise the river bed. Numerous flooding events have been traced to siltation of river channels. In addition, sediments can form up to 17% of floodwater volume (Calder and Awylward, 2006), increasing the height of peak water flows.
ii. *less water holding capacity in the soil* - forests increase water infiltration of the soil[23], allowing it to absorb and hold more water (higher levels of organic matter in forest soils can also further this).
iii. *faster movement of water across the land* - forests slow water as it flows over leaves and branches, leaf litter, as well as fallen trees and branches laying in small river channels (Nisbet and Thomas, 2006).
iv. *lower evapotranspiration* - forests intercept water and return it to the atmosphere.

Factors ii and iii above are thought to contribute to the 'sponge-effect' of forests (i.e. the hypothesis that forests absorb heavy rains and then slowly release this water over an extended period of time).

However, the science of forests and floods is still in flux. The sponge-effect hypothesis is controversial (see Enters et al., 2006). Research conducted mainly during the 1980s (mostly in the US, South America and Himalayas[24]) suggests that forests play a limited role in reducing flooding. Among hydrologists, the relationship between deforestation and large flood events remains controversial (Van Dijk et al., 2009). The consensus is that deforestation can have a role in flood formation for small/medium catchments (Tollan 2002). However, the significance of the role of deforestation in the formation of floods over larger catchments is disputed because these are influenced by a number of factors such as geological composition, topography and soil type (Reed 2002). Others contest this view and affirm that forests may in fact reduce even large scale floods (Alila et al., 2009). Analysis of (limited) data for

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[22] Semi-arid areas in Argentina include the driest parts of Patagonia and Western territory (Garbulsky and Deregibus, 2006).

[23] Soil permeability plays a critical role in flooding. Zimmermann et al. (2010) found that saturated hydraulic conductivity at 12.5 and 20 cm soil depth decreased by up to one order of magnitude after forest conversion to pasture, thus increasing overland water flow during heavy rains.

[24] See: Hewlett and Helvey (1970), and Hewlett and Bosch (1984), Gilmour et al. (1987), Hamilton (1987), and Kattelmann (1987).

larger catchments which have undergone either deforestation or afforestation (131 km^2 in Costa Rica and 94-1,545 km^2 in Chile) by Bathurst et al. (2011) suggests that the percentage change in forest cover must exceed 20-30% to record a measurable response in peak discharge.

Recent research (Ogden et al., 2013) provides some support to the sponge-effect hypothesis, at least in tropical forests. In an ongoing detailed field study in Panama, the amount of runoff from pastureland and forested land was measured following 450 storms (Ogden et al., 2013). It was found that even the largest storms on record (December 2010), which produced 520 mm of rainfall, did not overwhelm the ability of the forest catchment to store rainfall. During this large event the forest catchment produced about 35% less total runoff than a comparable partly deforested catchment (Ogden et al., 2013).

Tan-Soo et al. (2014) investigate the effect of deforestation on flooding in 31 river basins in Malaysia during 1984–2000 (a period when disaggregated land-use data are available as well as detailed data on flood events). Controlling for potentially confounding factors, they found that conversion of inland tropical forests to oil palm and rubber plantations significantly increased the number of flood days during the wettest months of the year.

Recent modelling of 'flood wave' travel time in relation to reforestation of floodplains has found that tree cover decreases flood peak (Dixon, 2014). If forests cover 10-15% of the total catchment area, then reductions of 5-6% in flood peak height can be seen after 25 years growth, with this reduction increasing to 7-8% after 50 years tree growth. If forests cover 20-35% of the catchment then they reduce flood peak height by 10-15% after 25 years of tree growth (with larger reductions after 50 years). This modelling only looks at the speed of water moving into and through the river network and does not take into account any reductions in the amount of water reaching rivers due to increased infiltration rates or evapotranspiration.

Whilst it is clear that forests can reduce peak water flow in some catchments (the size of catchment is disputed), other factors, such as topography, soil type/depth, and rainfall intensity (Van Dijk et al., 2009; Roa-García et al., 2011; Birkinshaw et al., 2011), will have a predominant role in determining flooding frequency. In addition, there appears to be a minimum level of forest cover for an appreciable reduction in flood peak. The scale of the benefit of flood control will depend on the location of the catchment in relation to human interests (i.e. downstream farmland, urban areas and infrastructure).

Argentina has been subject to major floods, resulting in economic losses as well as human casualties. Recent floods have been linked by some to deforestation (Finnerty, 2009; Staff, 2014), though other factors (such as climate change) may also play a role. High levels of deforestation of the Bosque Atlantico, through which major water bodies run, occurred between 1940 and the turn of this century (Di Bitetti *et al*. 2003). Based on the finding that the peak runoff rate from a pasture catchment is 1.7 times that of a tropical forest catchment (Ogden et al., 2013), one would expect that peak flow and hence flooding would have increased in the downstream area of the Bosque Atlantico as a result (the exact increase in runoff will depend on the soil and landscape characteristics of the areas deforested). Particularly damaging floods have recently occurred in Buenos Aires and neighbouring La Plata. Economic damages from flooding can also increase as a result of inappropriate development on floodplains, rather than an increase in flooding intensity.

Flood damage can hold back economic development, as well as causing financial hardship for individual households, industries and governments. The costs of the five most damaging floods in Argentina are shown in Table 19.

Table 19: Floods in Argentina 1900-2015 with damages of US$1 billion or more at year of event

Disaster	Date	Damage (US$)
Flood	April 2013	1 300 000 000
Flood	October 1985	1 300 000 000
Flood	April 1998	1 100 000 000
Flood	April 2003	1 028 210 000
Flood	May 1983	1 000 000 000

Source: EM-DAT (2016).

The results of floods in Tartagal, Salta. Greenpeace/Julio Pantoja

Forests can also sometimes moderate stream flows in the dry season, helping avoid severe droughts. However, the extent of droughts is disputed (Gilmour, 2014). Research, conducted mainly during the 1980s and 1990s, suggests that, especially in arid or semi-arid ecosystems, forests do not increase downstream water yield (Van Dijk and Keenan, 2007). Further research is required in this area. Ogden et al. (2013) in Panama found that flow from forested catchments receded more slowly than from mosaic and pasture catchments. The runoff rate from the forest catchment was 1–50% greater than that from a mosaic catchment of the same size at the end of the dry season, supporting the sponge-effect hypothesis (at least in tropical forests) (Ogden et al., 2013). As for flood control, factors such as topography, geology and rainfall patterns will play a key role in determining the exact relationship between forests and stream flows (Bruijnzeel, 1990).

In addition, tropical forests can act as a giant 'water pump', releasing billions of litres from the trees into the air in the form of vapour. A large tree with a crown 20 metres across evaporates up to 300 litres a day (Rocha, 2014). The resulting clouds or 'flying rivers' (the term coined by Antonio Nobre) bring rain to other areas and without them, the area that produces 70% of South America's GDP would be desert according to Nobre (Rocha, 2014). As Nobre is quoted as saying, "Of course, we need agriculture, but without trees there would be no water, and without water there is no food. A tonne of soy takes several tonnes of water to produce… Water is the main agricultural input." (Rocha, 2014).

Deforestation has been blamed for the water shortage being experienced in the central Argentine province of Córdoba (Frayssinet, 2013). Only 5% of the 12 million hectares of native forest that the province had at the start of the 20th century remain, and as a result some towns have had to ration water (Frayssinet, 2013). Drought has a less visible impact on economic output than floods, but it is not a negligible one. Crop yields are reduced and certain economic activities are curtailed, the cost of water as an input to production increases, and investments in alternative water sources are required. The cost of drought in Argentina has been estimated at US$60 million per event (EM-DAT, 2014) (though the role of deforestation will only be one small element of this).

Forest biodiversity and landscapes as a tourism resource

Forests can provide a desirable tourist destination, based on the scenic landscape qualities as well as the rare animal species they harbour. The second most frequented destination for tourists in Argentina is the Iguazu Falls, and eco-lodges have been established in Iguazu's subtropical forest.

Iguazu Falls, Argentina, by chensiyuan, Wikimedia Commons.

Tourism brings billions of dollars annually into Argentina and tourist numbers have been steadily increasing over recent decades (Secretaría de Turismo de la Nación y Consejo Federal de Inversiones, 2005). Between 2001 and 2010 the number of international tourists visiting Argentina doubled (World Bank, 2014a), whilst visitor numbers to National Parks increased four-fold during this period (Figure 6). Many countries have demonstrated that wildlife and eco-tourism can be profitable, and in fact eco-tourism has a long history in Argentina. However, as a growing market there is likely to be untapped potential and a number of possible high-value eco-tourist destinations may have been lost as a result of deforestation in past decades. Quantifying the lost opportunity is beyond the scope of this study.

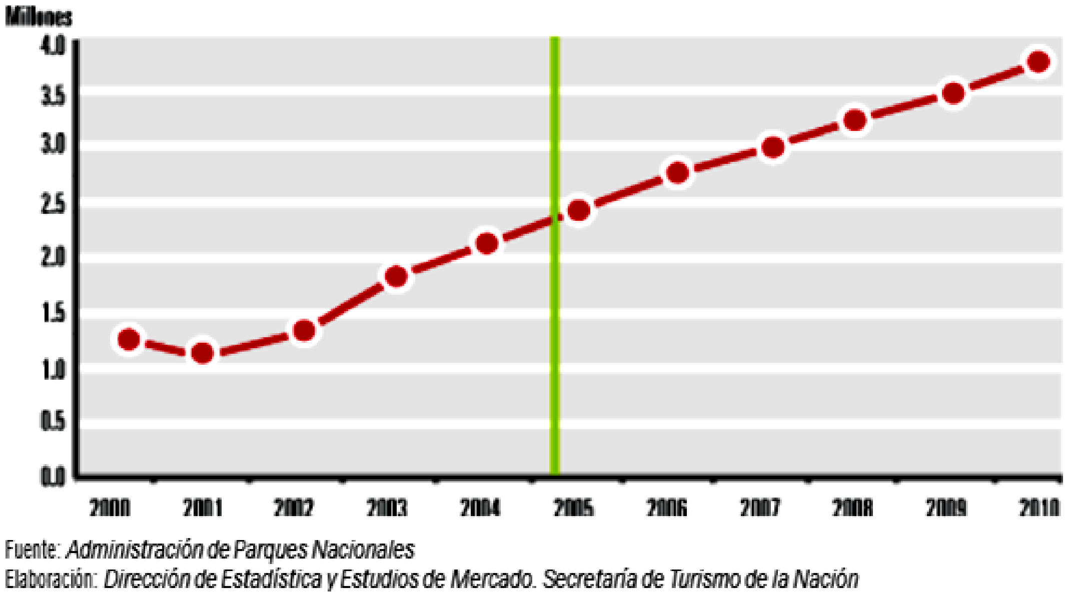


Figure 6: Tourist visits (million per year) to National Parks in Argentina

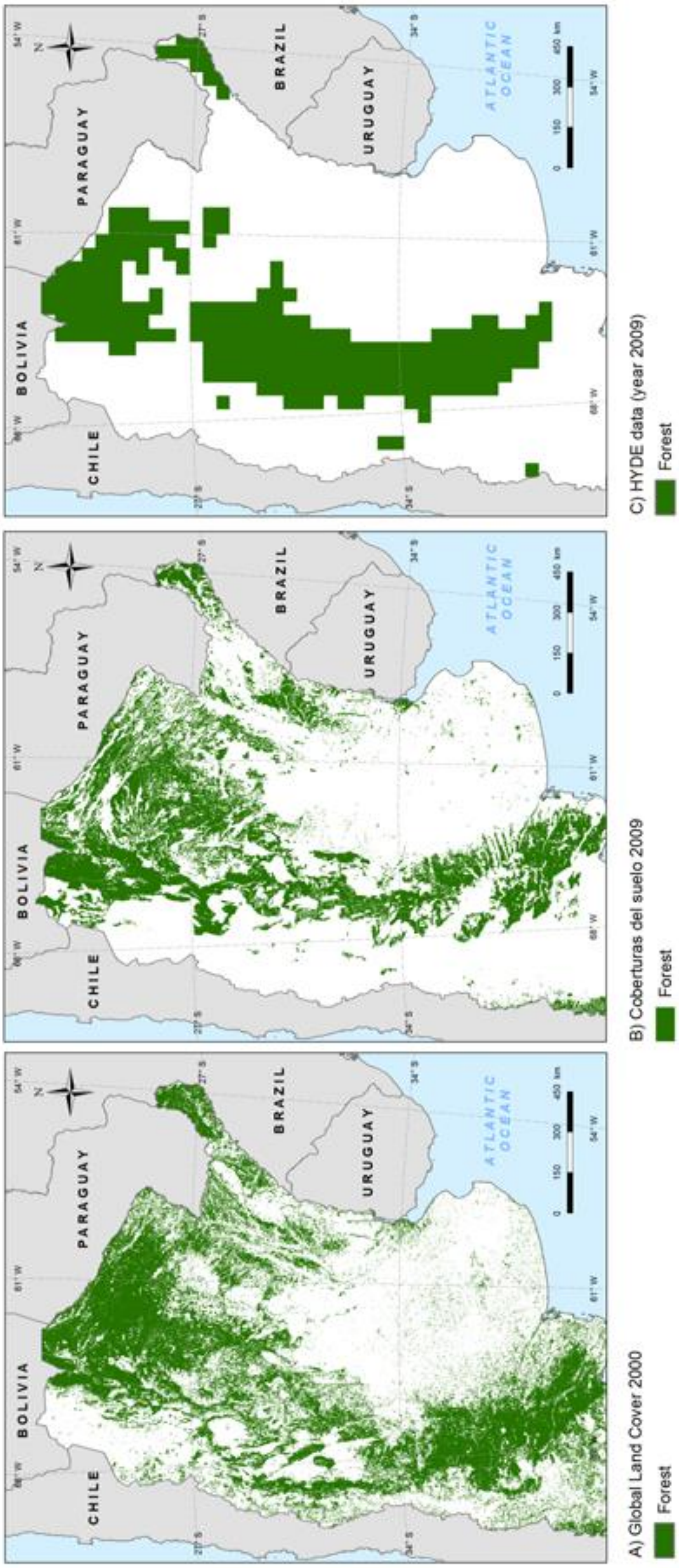
6.6 Estimation of quantity and value of different services lost due to deforestation

To estimate losses from deforestation, it is necessary to identify both the area of land-use change and its location. Detailed maps of national forest cover going back to 1960 are not readily available and so forest cover is here estimated for the earliest decades. It was decided to focus on northern Argentina, where the greatest extent of natural forest exists and where deforestation pressures have been greatest. In order to model forest cover, the HYDE data set was utilized; it uses population density to estimate the spatial and temporal distribution of land use from 1700 to 2000 (Goldewijk, 2001).

Although the HYDE dataset is relatively coarse (50 km x 50 km grid squares) it should nevertheless pick up the largest areas of forest. In order to explore the reliability of HYDE data at this resolution for mapping forests in northern Argentina, the mapped HYDE data was put side by side with two other datasets roughly comparable in terms of the time period that they represent (Figure 7). The national dataset was for the year 2009, and the global dataset used was GLC 2000. The HYDE data for 2009 was used as a comparison. Whilst HYDE is missing some areas of forest and over-emphasizes the size of others, the broad pattern of forest cover is replicated.

Therefore, the HYDE data was used to estimate historic forest cover in Argentina (Figure 8-12). Land uses were grouped into broad habitat headings, with three headings being of particular interest: crop, pasture, and forest. Land-use change was modelled over each decade from 1960 to 2010 (see Appendix B for more detail on the methodology).

The impact of this change in forest cover on the quantity of the ecosystem services being delivered was estimated for each of the services listed on page 32.



Methods and data sources:

A) Global Land Cover 2000 database. European Commission, Joint Research Centre, 2003. **http://bioval.jrc.ec.europa.eu/products/glc2000/glc2000.php**

Forest cover was considered in this map to include the following land cover categories: Tree Cover, broadleaved, deciduous, closed; Tree Cover, broadleaved, deciduous, open; Tree Cover, broadleaved, evergreen; Tree Cover, mixed leaf type; Tree Cover, regularly flooded, fresh water; Tree Cover, regularly flooded, saline water; Shrub Cover, closed-open, deciduous; Shrub Cover, closed-open, evergreen.

B) Coberturas del Suelo 2009. El Instituto Geográfico Nacional de la República Argentina. **http://www.ign.gob.ar/sig**

Forest cover was considered in this map to include the following land cover categories: Bosque artificial; Bosque en galería; Bosque quemado; Bosque, selva, foresta, parque natural intransitable; bosque, selva, foresta, parque natural transitable; Coniferas; Palmar; Vegetacion leñosa.

C) Meiyappan, P., and Jain, A. K. (2012). Three distinct global estimates of historical land-cover change and land-use conversions for over 200 years. Frontiers of Earth Science, 6(2), 122-139. DOI: 10.1007/s11707-012-0314-2. See: **http://climate.atmos.uiuc.edu/ISAM_Landuse/land-cover_doc_c20130831.pdf**. The Historical Land-Cover Change and Land-Use Conversions Global Dataset used in this study were acquired from NOAA‟s National Climatic Data Center(**http://www.ncdc.noaa.gov/**). Presented here is the data for the year 2009. Forest cover was considered in this map to include the following land cover categories: Tropical Evergreen Broadleaf Forest; Tropical Deciduous Broadleaf Forest; and Shrubland.

Figure 7: Comparison of land cover datasets

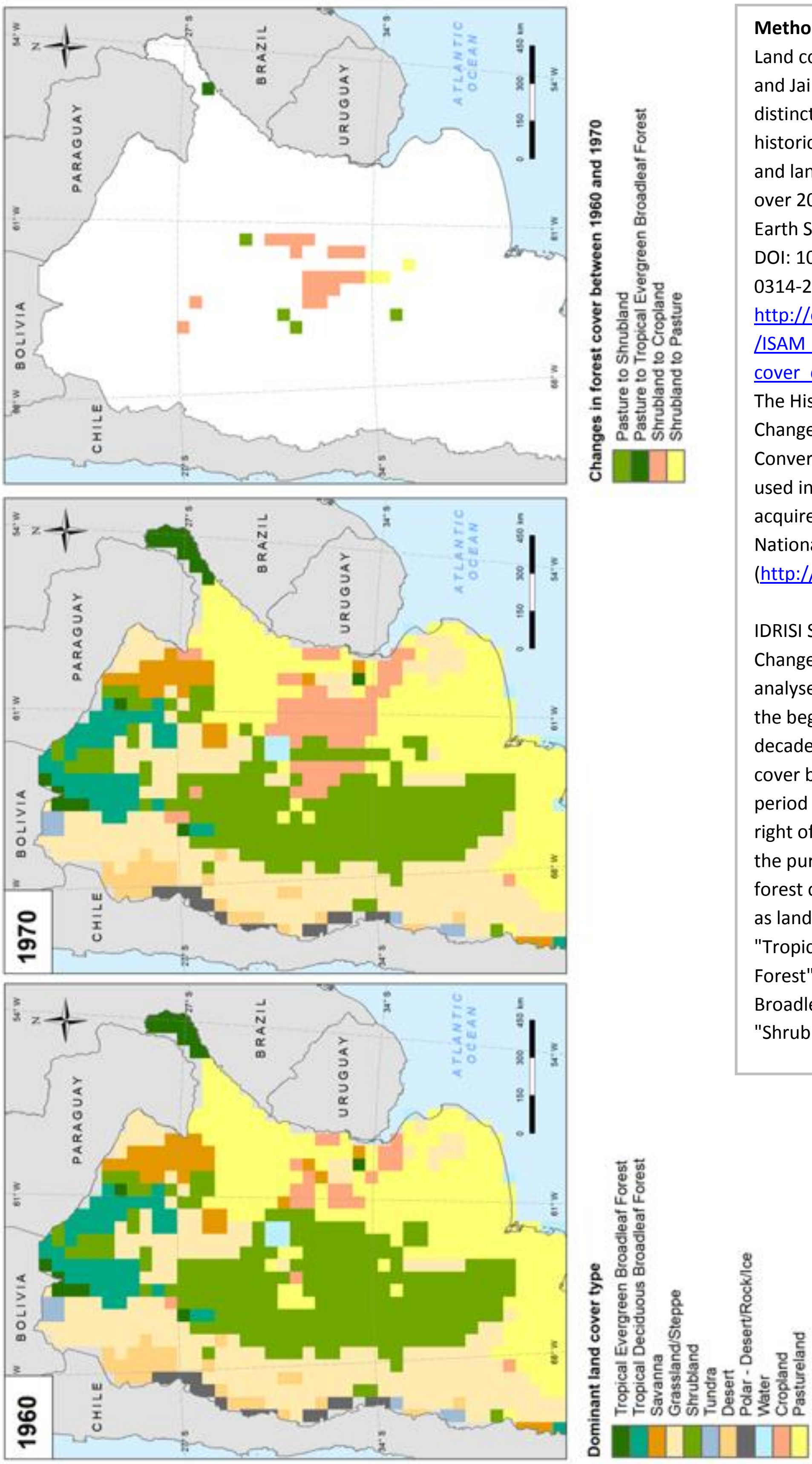
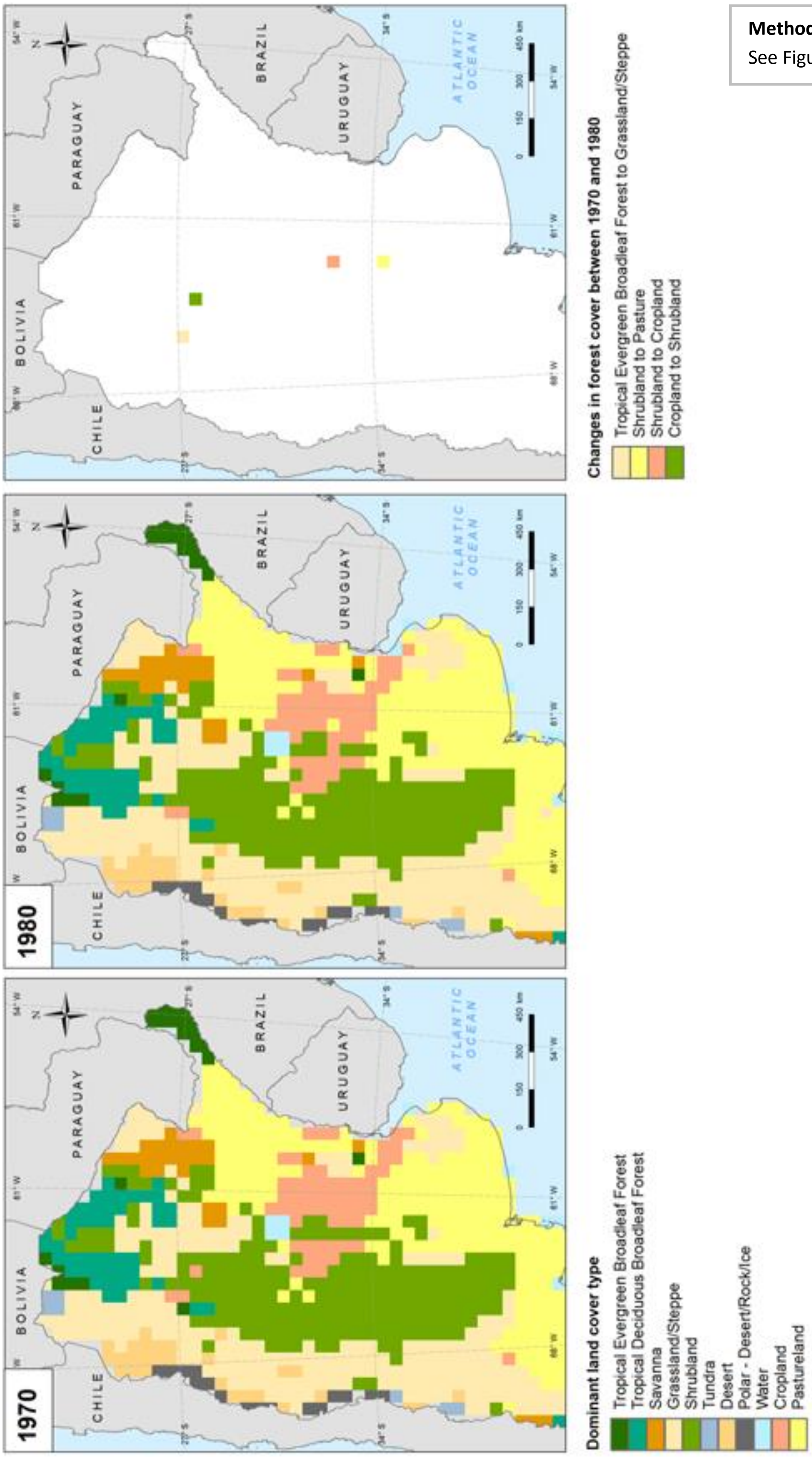


Figure 8: HYDE data for land-use change 1960 to 1970

Methods and data sources:
Land cover: Meiyappan, P., and Jain, A. K. (2012). Three distinct global estimates of historical land-cover change and land-use conversions for over 200 years. Frontiers of Earth Science, 6(2), 122-139. DOI: 10.1007/s11707-012-0314-2. See: http://climate.atmos.uiuc.edu/ISAM_Landuse/land-cover_doc_c20130831.pdf. The Historical Land-Cover Change and Land-Use Conversions Global Dataset used in this study were acquired from NOAA‟s National Climatic Data Center (http://www.ncdc.noaa.gov/).

IDRISI Selva 17.0 Land Use Change Modeller was used to analyse land cover data for the beginning and end of each decade. The changes in forest cover between this time period are shown on the far right of each time period. For the purpose of this study, forest cover has been classed as land cover categories "Tropical Evergreen Broadleaf Forest"; "Tropical Deciduous Broadleaf Forest"; and "Shrubland".



Methods and data sources:
See Figure 9.

Figure 9: HYDE data for land-use change 1970 to 1980

Methods and data sources:
See Figure 9.

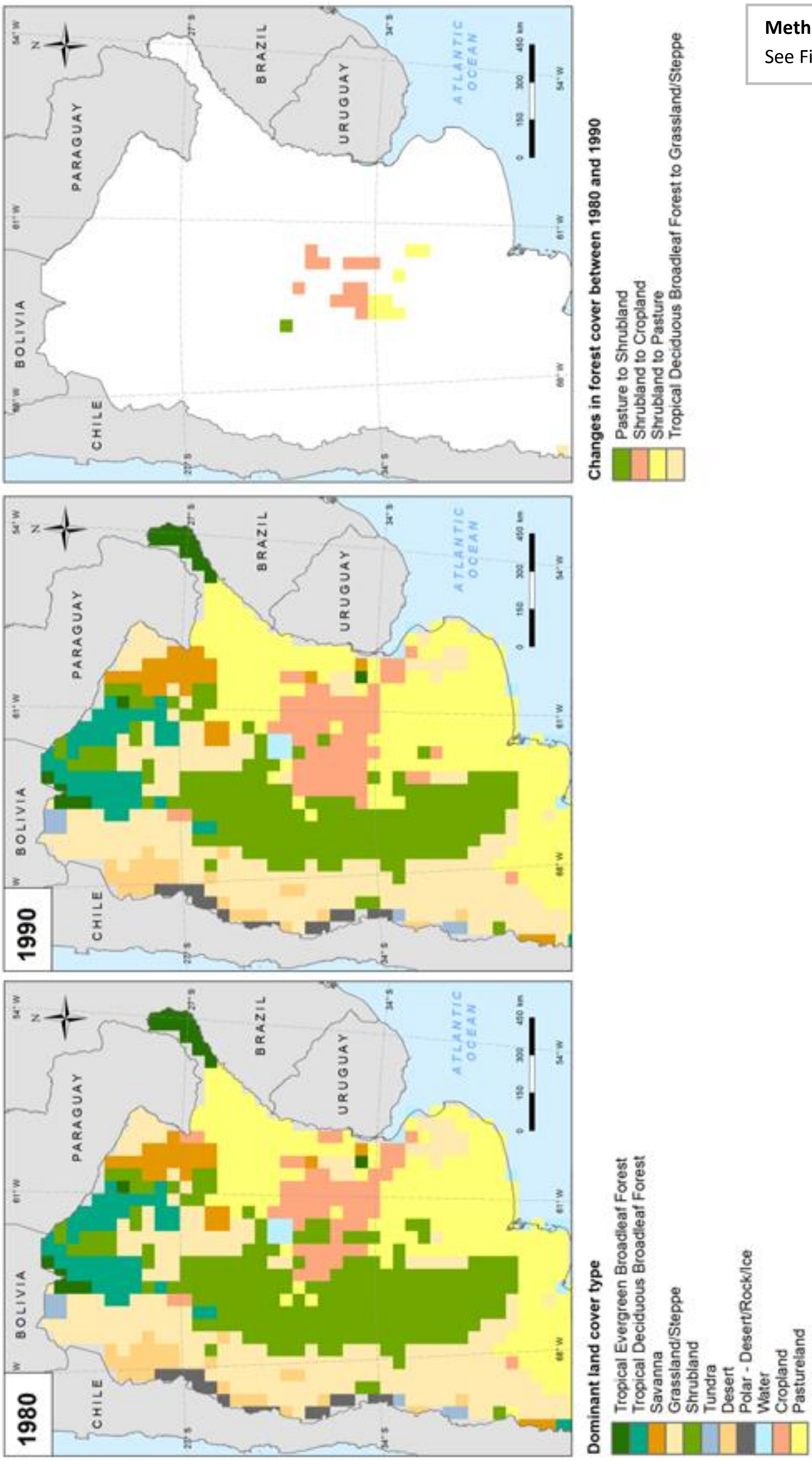


Figure 10: HYDE data for land-use change 1980 to 1990

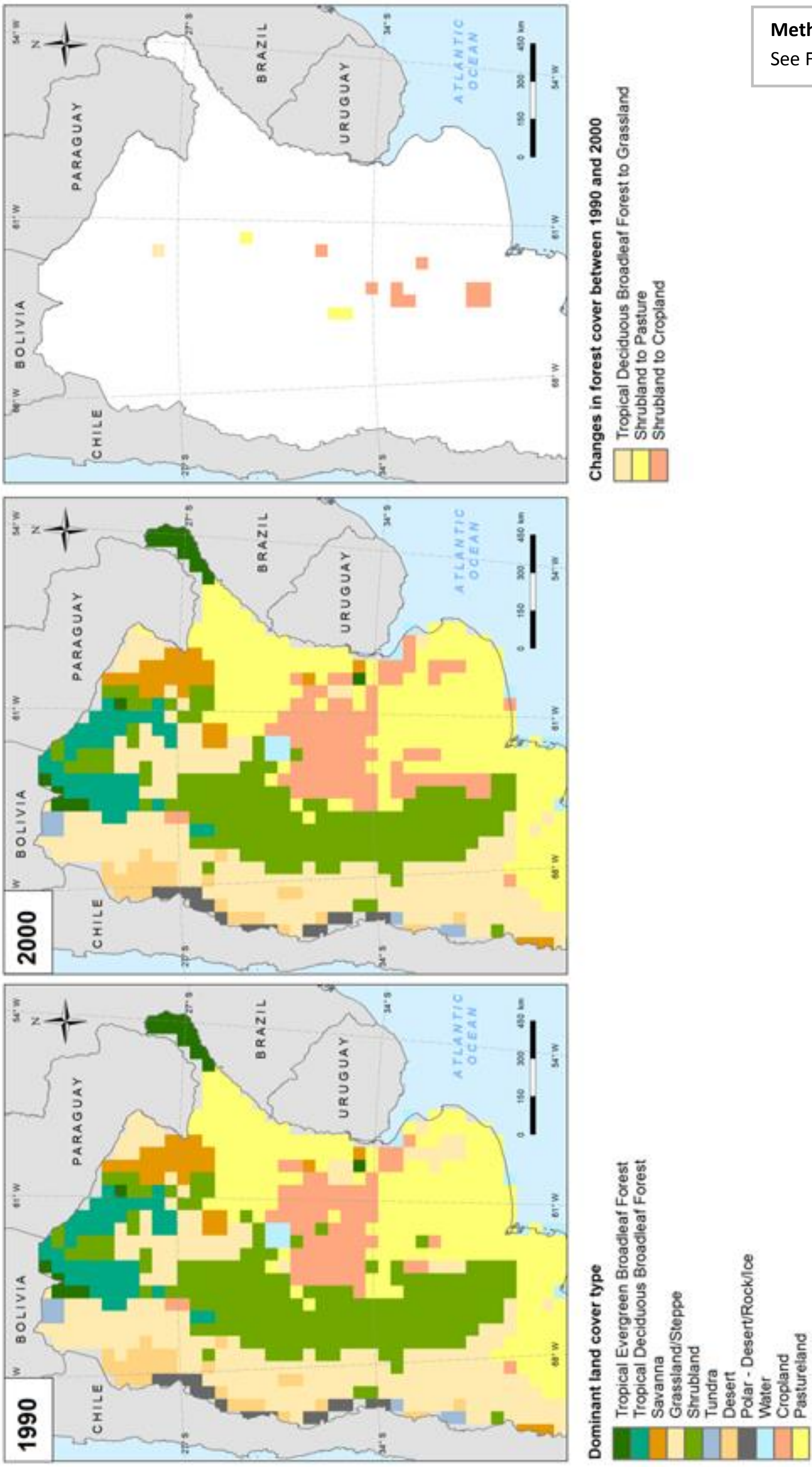


Figure 11: HYDE data for land-use change 1990 to 2000

Methods and data sources:
See Figure 9.

Methods and data sources:
See Figure 9.

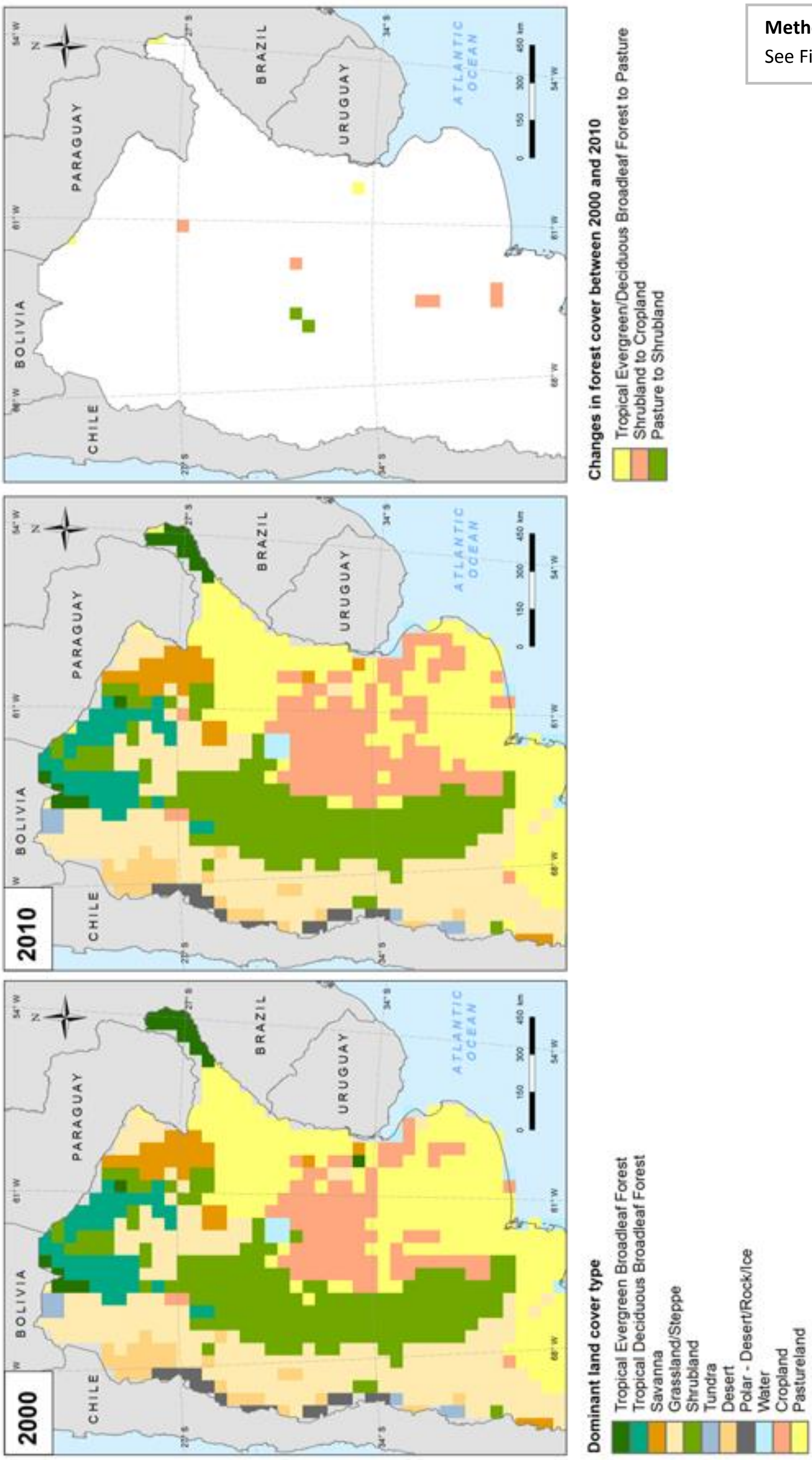


Figure 12: HYDE data for land-use change 2000 to 2010

Following the quantification of ecosystem service losses, the value of each service is determined on a unit basis and then multiplied by the quantity in order to estimate the total value of loss. This analysis is carried out for each of the eight ecosystem services identified to estimate impacts of deforestation from 1960 to 2010, up to the year 2099.

Figures for the present value are presented for two discount rates, 5% and 1%, (Box 2; the impact of discount rates on the results is discussed in section 6.7). The estimation of future values for sustainable wood products, NWFPs and other forest ecosystem services until the end of this century is inherently a task involving a great deal of uncertainty. However, the same applies to the crop values used. The alternative is to only value services in the decade being analyzed. However, since the impacts of climate change are expected to occur over the whole of this century (and beyond) this would ignore one of the major costs of deforestation.

Box 2. Discounting

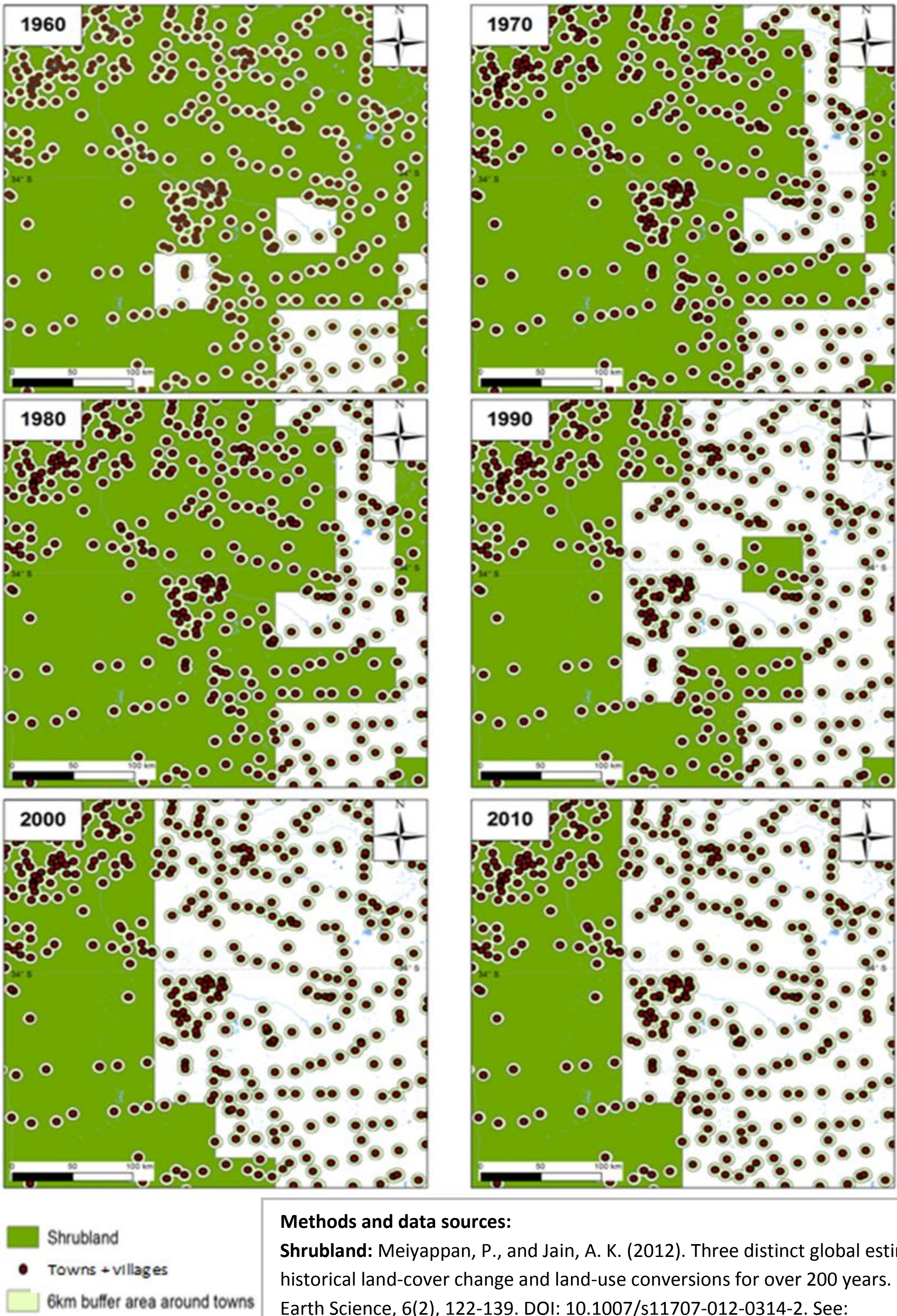
Discounting is a technique used to compare costs and benefits that occur in different time periods. It is a separate concept from inflation, and is based on the principle that, generally, people prefer to receive goods and services now rather than later. This is known as 'time preference'. Any preferences for sustainable supply of goods and services are ignored using this approach. For individuals, time preference can be measured by the real interest rate on money lent or borrowed (i.e. interest rate minus inflation rate, thus the interest 'above' inflation). The mathematical expression used to calculate the discounted present value (PV) of an amount of money (N) in a future time period (i.e. 't' number of years) is:

$$PV = \frac{N}{(1+r)^t}$$

where r is the chosen discount rate. The discount rate converts all costs and benefits to 'present values', so that they can be compared. A common choice of discount rate is 5% (i.e. r = 0.05). For example, a payment of $105 one year from now has a present value of $100 (as if you put $100 in the bank now at a 5% interest rate, next year it would be worth $105). Calculating the present value of the differences between the streams of costs and benefits over the period of a time planning horizon (e.g. 25 years) provides the *net* present value (NPV) of an option, which is the primary economic criterion for deciding whether an investment is worthwhile.

The basic principles of discounting future cash flows were addressed by English economists almost 400 years ago, and further detailed by German forest economists in the early 19th century. However, with regard to discussions on sustainable development, over recent decades there has been debate amongst environmental economists about the discount rate which should be used when assessing long-term investments. Where impacts relate to society rather than individuals, and where there are potentially large risks and associated uncertainty of occurrence, then a lower discount rate has been advocated. This is because a high discount rate effectively ignores large impacts that occur sometime in the more distant future, favouring immediate benefits. Stern (2007) used a low discount rate of 1.4%, advocating the adoption of a low rate of time preference, others have argued for a negative discount rate for climate change policies (Fleurbaey and Zuber, 2013) (see Appendix A for further discussion of discount rates in relation to climate change calculations).

6.6.1 Harvested NWFPs



Methods and data sources:
Shrubland: Meiyappan, P., and Jain, A. K. (2012). Three distinct global estimates of historical land-cover change and land-use conversions for over 200 years. Frontiers of Earth Science, 6(2), 122-139. DOI: 10.1007/s11707-012-0314-2. See: http://climate.atmos.uiuc.edu/ISAM_Landuse/land-cover_doc_c20130831.pdf. The Historical Land-Cover Change and Land-Use Conversions Global Dataset used in this study were acquired from NOAA"s National Climatic Data Center (http://www.ncdc.noaa.gov/).
Towns: El Instituto Geográfico Nacional de la República Argentina. Centros poblados (2013). Downloaded from: http://www.ign.gob.ar/sig October 2014

Figure 13: Forest area (green) and NWFP harvest areas around villages and towns each decade, sample area for illustration

The lost NWFP harvest area was calculated based on a 6km accessibility buffer around villages and towns (Figure 13). The size of the buffer could be varied by type of NWFP using more complex models (Schaafsma et al., 2012) (Widayati et al., 2010) to improve accuracy.

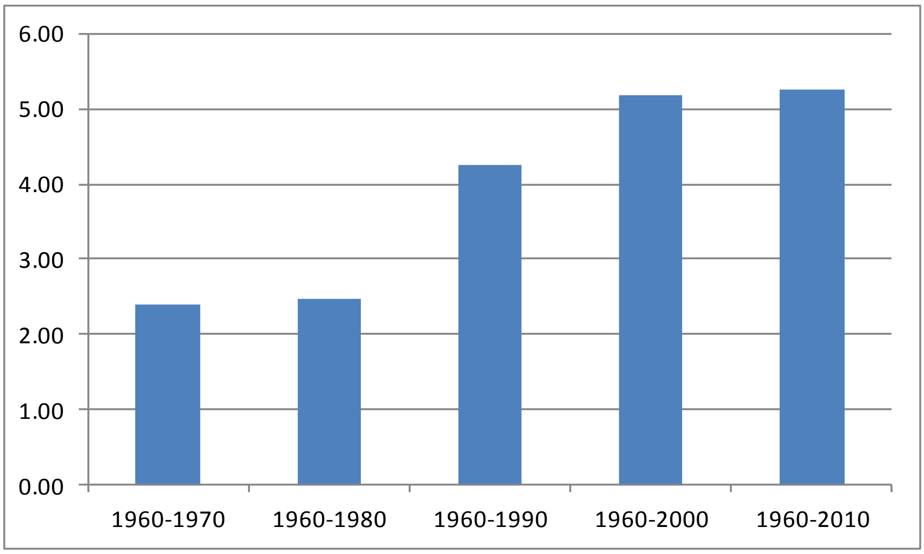


Figure 14: Cumulative loss of NWFP harvest areas in forests around villages and towns (millions of hectares).

To value NWFPs based on these estimated losses, an average value per hectare is required. The total value of NWFPs currently marketed in Argentina (Table 13) divided by the current total forest area gives a value of US$4 per ha per year. However, this underestimates the per hectare value since many areas of forest have no NWFPs collected, thus in reality the total value relates to an area smaller than the total forest. In addition, many collected NWFPs are consumed directly rather than marketed. Only the most accessible forest area is included in this analysis: beyond 6 km from the population site, the forest is classed as inaccessible and an NWFP value of zero is assigned. Hence, a higher NWFP value than US$4 per ha per year should be used. Meta-analyses of forest values indicate that the values could be as high as US$200-300 per ha per year (de Groot et al. 2012). For this analysis, an NWFP value of US$20 per ha is adopted (constant value, 1960 to 2099). This value is multiplied by the area to give a total value (Table 20).

Table 20: Value of lost NWFPs (US$) from year of loss up to 2099 from deforestation in each decade, 1% and 5% discount rate

Decade of loss	5%	1%
1960-70	3,345,979,715	5,217,726,980
1970-80	86,861,778	143,586,121
1980-90	1,767,357,232	3,153,553,485
1990-2000	742,420,342	1,471,254,024
2000-2010	39,613,513	91,578,317

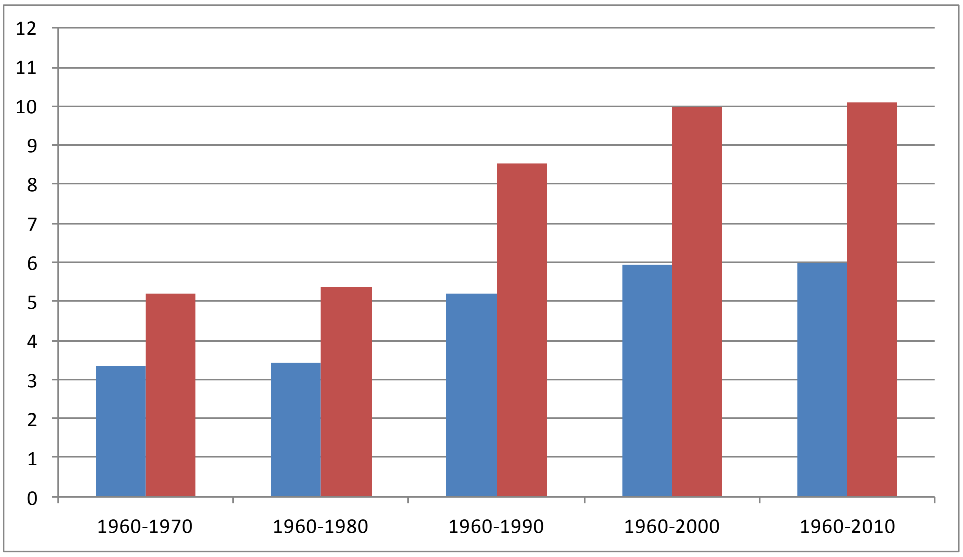


Figure 15: NWFP cumulative losses to 2099 from deforestation each decade (2010 US$ billion), (5% discount rate - blue bars, 1% discount rate - red bars).

6.6.2 Sustainably harvested wood

Accessible areas were identified (6km buffer around villages and towns)[25]. Even with annual growth increments of 25m^3/ha per year possible in tropical forests (Pandey, 1990), the estimate of usable natural forest yield is only about 1 m^3/ha per year of commercial species (Odokonyero, 2005). For dry forests (shrubland) it is assumed that the sustainable harvest rate would be much less (IPCC, 2003), so that harvested wood available for sale is just 0.165m^3/ha per year (Figure 16).

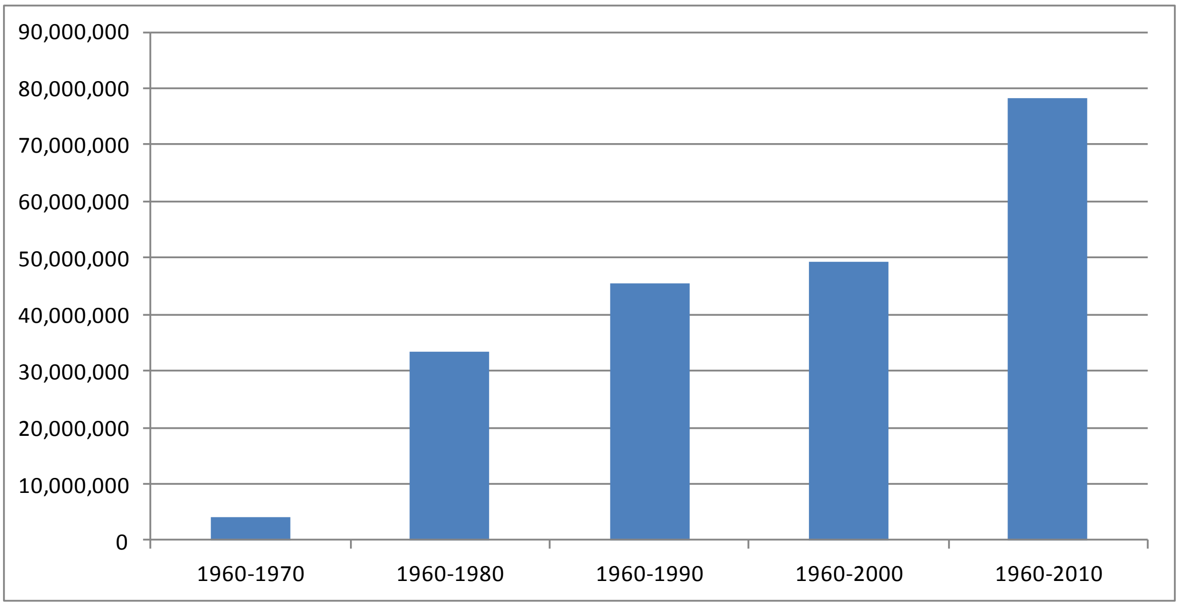


Figure 16: Cumulative loss of sustainable wood harvest potential (m^3)

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[25] This is a very conservative assumption, since it ignores access from roads running between villages and towns.

Real (year 2010) prices for timber going back to 1960 were obtained from the World Bank Commodity Price Data (The Pink Sheet), and these were adjusted based on the price difference between the World Bank series and the timber price in Argentina in 2005 (Cubbage et al., 2007). Going forward to 2050, a 0.6 % annual price increase was used, which is below the historic increase of 1.9 % over the last 50 years (Haynes, 2003). However, there are also long-term forecasts of resource shortages (Lee et al., 2012), and persuasive arguments that in a world rapidly reducing carbon emissions, the price of wood will reflect both a greater importance as a carbon-neutral fuel source and an increasing scarcity as unsustainable wood product supplies are phased-out. Therefore, for 2051-2099, a higher rate of half the historic annual increase of 1.9%, i.e. 0.95%, is used.

Table 21: Value of lost sustainable wood harvesting (US$) from year of loss up to 2099 from deforestation in each decade, 1% and 5% discount rate

Decade of loss	5%	1%
1960-70	784,626,039	1,339,030,212
1970-80	4,663,350,706	9,164,007,830
1980-90	1,690,601,576	3,557,132,987
1990-2000	437,926,436	1,002,804,980
2000-2010	2,520,653,702	6,959,784,413

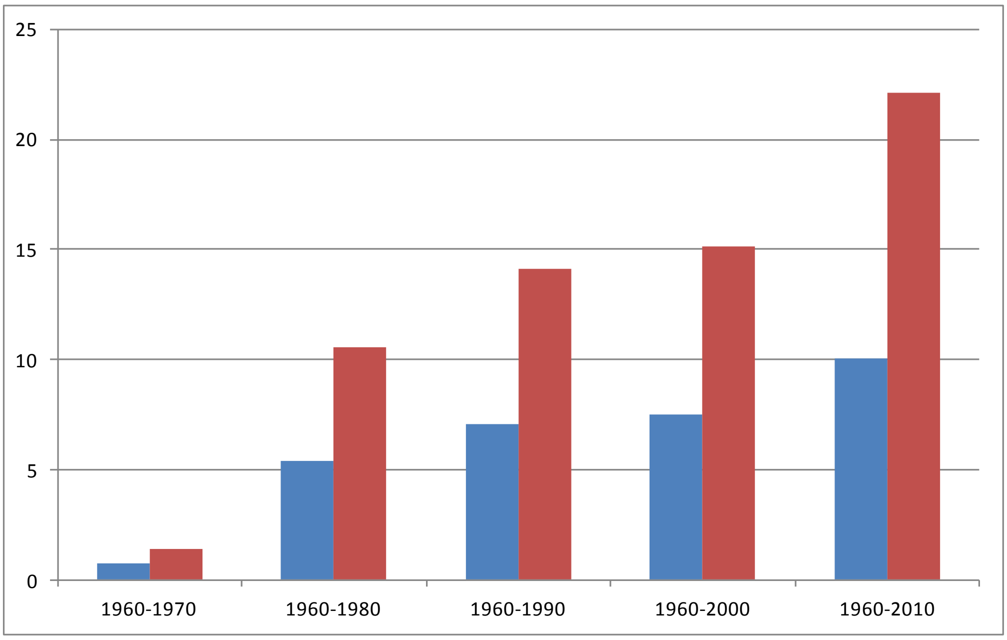


Figure 17: Sustainable wood products cumulative losses to 2099 from deforestation each decade (2010 US$ billion), (5% discount rate - blue bars, 1% discount rate - red bars)

6.6.3 Pollination of crops bordering the forest

In order to identify where there are pollination benefits to crops from forests buffering cropland, relatively detailed spatial information is required, as not all crops benefit from pollination. The HYDE data used in the analysis was judged to be too coarse to allow an analysis of pollination benefits of an acceptable quality, as it is relatively large-scale and uses majority land use in its categorizations. Whilst there is a general trend of loss between 1970 and 2010, pollination benefits were not examined further in this analysis.

6.6.4 Regulation of diseases (to humans)

Based on the major diseases associated with deforestation that do/could impact Argentina the total number of estimated cases for these in 2000-2010 in Argentina was estimated as the number of American Cutaneous Leishmaniasis (ACL) and dengue cases (identified in section 6.5), and increased by 50% to account for other diseases (based on the relative incidence of diseases in the region). However, it was not assumed that all of these cases ultimately originate from deforestation. Research in Brazil has found that a 4% rise in deforestation results in a 48% increase in rates of malaria (Olson et al., 2010). It is assumed that a linear relationship exists and thus disease rates were 48% higher in Argentina in 2000-2010 than without deforestation (since in 2000-2010 4% of the forest area was converted). Therefore, on average around 13,400 cases per decade emerge where 2.5 million hectares of forest (4% of the total) are lost. This is equivalent to 1 case per 186 hectares, and this rate is used for previous decades (Table 22). Obviously, detailed epidemiological studies would provide a better basis for estimating disease rates.

Table 22: Number of disease cases each decade associated with deforestation

	Cases
1960 - 1970	30,914
1970 - 1980	4,032
1980 - 1990	29,570
1990 - 2000	18,817
2000 - 2010	13,441

In the above, it is assumed that disease increases will only occur in the decade in which the deforestation took place, however, this may be underestimating the risk. Forest fragmentation changes host and vector abundance and distribution, and thus the dispersal of pathogens (Wilcox and Ellis, 2006). For instance, disturbed habitats with low-diversity present the greatest disease outbreaks risk in relation to rodent-borne haemorrhagic fevers (Millsa, 2006). Given the possibility of adaptation of disease vectors to modified human landscapes (Salomón et al., 2012) it is therefore assumed (conservatively) that 5% of the deforested area will be at risk of vector adaptation leading to a disease prevalence rate equivalent to the decadal border-effect. Whilst not included here, there may also be economic losses associated with increased diseases amongst livestock.

The cost of disease is estimated using the average treatment cost (US$194 for treatment of ACL with miltefosine[26]) plus loss of income (of US$125 per person in Puerto Rico due to dengue) (Suaya et al., 2006) as representative costs of the relevant diseases, multiplied by the number of disease cases each decade associated with deforestation.

[26] A combination of national cost data for drugs and the health care staff costs would be preferable.

Table 23: Estimated cost of disease (US$) within each decade associated with deforestation

	Cost per decade
1960 - 1970	9,861,559
1970 - 1980	1,779,368
1980 - 1990	9,521,764
1990 - 2000	6,478,776
2000 - 2010	4,611,573

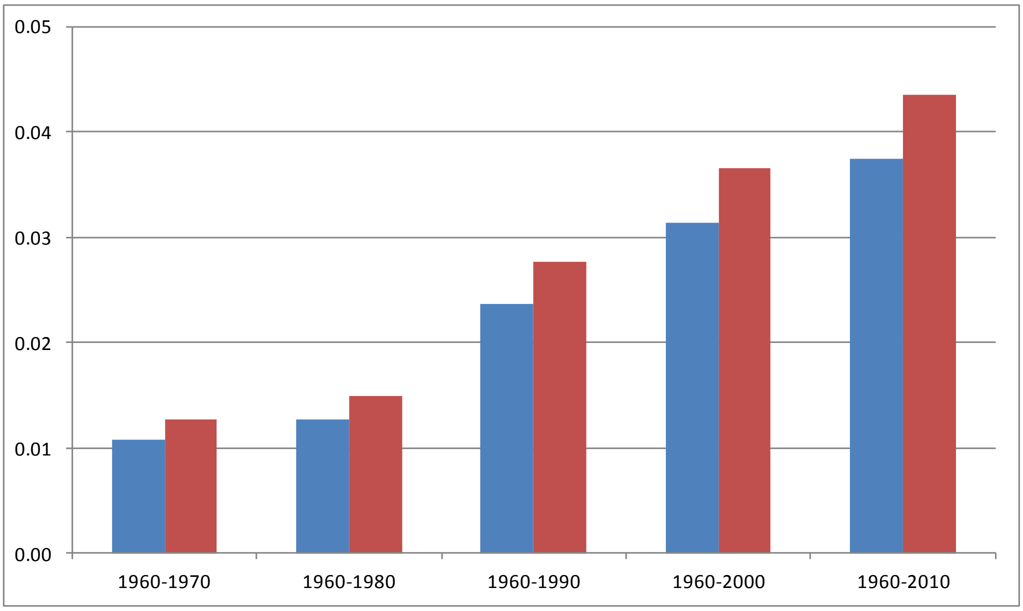


Figure 18: Cumulative disease losses to 2099 from deforestation each decade (2010 US$ billion), (5% discount rate blue bars, 1% discount rate red bars)

Clearly the disease cost analysis could be improved, but since the costs are relatively small in this instance it would appear that expending further time and effort to improve these estimates would not be worthwhile. However, for other countries, disease costs could be a significant factor to be considered, and hence more detailed analysis would be required.

6.6.5 Carbon retention/sequestration

The carbon losses associated with different land-use changes were estimated as follows (see Appendix B for more detail): an above-ground carbon content of 5 t/ha was assigned to cropland (rather than zero tonnes, as in the IPCC Tier I approach), and a content of 19 t/ha was assigned to pasture. The land-use changes yielded the carbon loss estimates in Table 24.

Table 24: Carbon losses per hectare from changes in land uses

Carbon losses from...	t C / Ha
SHRUB to Cropland	33.2
SHRUB to Pasture	19.5
TEBF to Grassland	88.4
TDBF to Grassland	52.6
TEBF to Pasture	109.5
TDBF to Pasture	73.7

Note: SHRUB = Shrubland, TEBF = Tropical Evergreen Broadleaf Forest, TDBF = Tropical Deciduous Broadleaf Forest

Based on the land-use changes seen in the HYDE data, and additional soil organic carbon losses (top metre of the soil profile), Gasparri et al. (2008), decadal losses of carbon from deforestation were estimated as in Table 25.

Table 25: Total carbon lost per decade

Decade	Total C lost (t)
1960 - 1970	250,073,870
1970 - 1980	41,628,097
1980 - 1990	216,164,096
1990 - 2000	149,392,277
2000 - 2010	175,969,339

To evaluate these carbon estimates, a comparison was made with another analysis of deforestation and biomass carbon losses, which used a different methodology (Gasparri et al., 2008) (though the same carbon values per hectare for soils are used). Gasparri et al. (2008) estimate that deforestation in northern Argentina released 209 million tonnes of carbon between 1996 and 2005, which compares with 163 million tonnes using the HYDE data. However, whilst lower for this period, over the period 1960 to 2010 as a whole the HYDE data is at the upper confidence interval of the Gasparri et al. (2008) estimates. Therefore, the estimates appear plausible.

The valuation of carbon involves consideration of the costs that excess carbon in the atmosphere will bring about. Argentina is vulnerable to the impacts of climate change, as well as being responsible (along with the other nations of the world) for the causing it. A social cost of carbon (SCC) is estimated to provide a variable cost of carbon emissions for each decade from 1960 - 2010. Two approaches are adopted, using different discount rates: one based on the US EPA's SCC estimate and the other using an alternative approach (see Appendix A for details on the SCC methodology adopted). Carbon values are applied to the carbon emissions resulting from deforestation in northern Argentina each decade from 1960 to 2010 (US$ 0.96, 1.61, 2.68, 4.48, 7.49 t/CO_2 respectively for each decade using a 5% discount rate, and US$ 17.20, 20.33, 24.85, 38.30, 40.49 t/CO_2 using 1%). This results in the estimate of decadal costs (or 'carbon debt') shown in Table 26.

Table 26: The SCC for each decade using two different approaches

US EPA (5% discount rate)		Alternative (1% discount rate)	
Decade	Cost US$	Decade	Cost US$
1960 - 1970	882,760,759	1960 - 1970	15,784,662,644
1970 - 1980	245,605,773	1970 - 1980	3,105,872,322
1980 - 1990	2,129,216,348	1980 - 1990	19,714,165,573
1990 - 2000	2,458,996,874	1990 - 2000	20,998,578,411
2000 - 2010	4,837,397,119	2000 - 2010	26,149,043,719

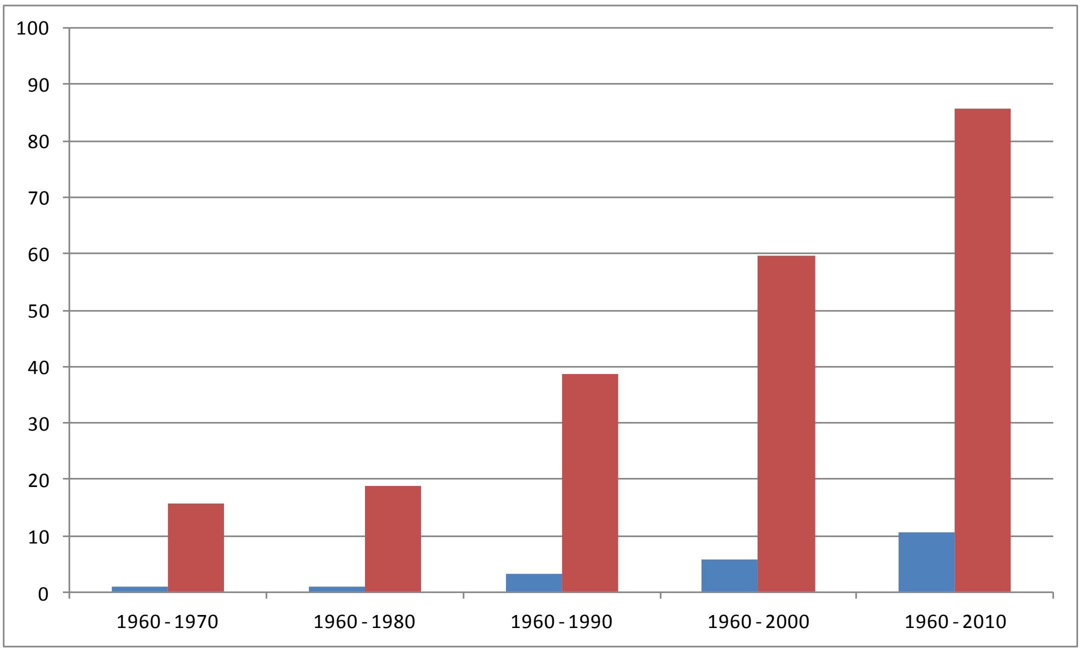
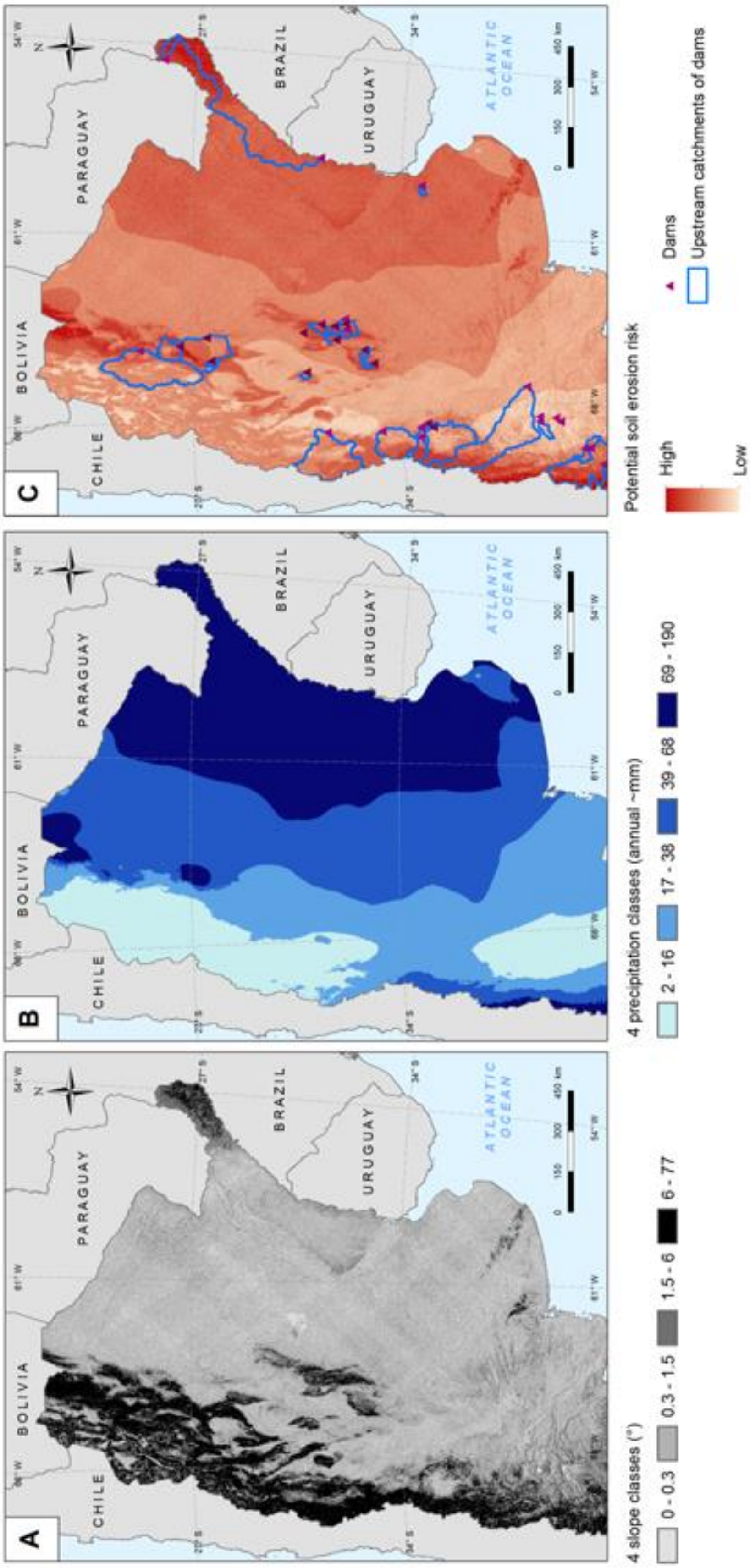


Figure 19: The cumulative carbon costs (SCC) in 2010 US$ billion (5% discount rate (EPA) - blue bars, 1% discount rate (Alternative) - red bars)

6.6.6 Reduced sedimentation through limiting soil erosion

As forests provide protection against soil erosion, it is possible to identify soils at risk of erosion by mapping deforestation in river basins (Figure 20). Using the HYDE data on forest loss for each decade, along with data on dams (Lehner, 2008b), we identified dam catchments that had undergone full or partial deforestation (Figure 21)[27] in the decades when the dam was built and immediately afterwards. It was assumed that on average 74 tons (67 tonnes) of soil would be lost over a decade for each hectare of forest cleared (Crafford et al., 2012, Figure 22).

[27] Sediments also have impacts on fisheries and freshwater supplies. However, it was not possible in this study to consider these further.



Methods and data sources:
The relative potential risk of an area has been evaluated as a function of slope and rainfall. This method uses an overlay approach, (A + B = C) where annual average precipitation per cell (split into 4 classes using a quantile classification) has been combined with data generated for slope (split into 4 classes using a quantile classification). Since there are 4 classes for both slope and precipitation, the resulting output (C) has a maximum value of 8 and a minimum value of 2. The classes represent a low to high potential for erosion risk. Higher values represent higher erosion impact in the absence or degradation of forests. No weighting of the importance of the different input factors is used in this simple approach. For details of the methodology used to create the upstream dam catchments, see Figure 22.

Elevation: Lehner, B., Verdin, K., Jarvis, A. (2008): New global hydrography derived from spaceborne elevation data. Eos, Transactions, AGU, 89(10): 93-94. See: http://hydrosheds.cr.usgs.gov/. **Precipitation:** WorldClim http://www.worldclim.org/. **Dams:** Lehner, B., R-Liermann, C., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P. et al.: High resolution mapping of the world's reservoirs and dams for sustainable river flow management. Frontiers in Ecology and the Environment. Source: GWSP Digital Water Atlas (2008). Map 81: GRanD Database (V1.0). Available at http://atlas.gwsp.org.

Figure 20: Mapping slope, precipitation and soil erosion risk

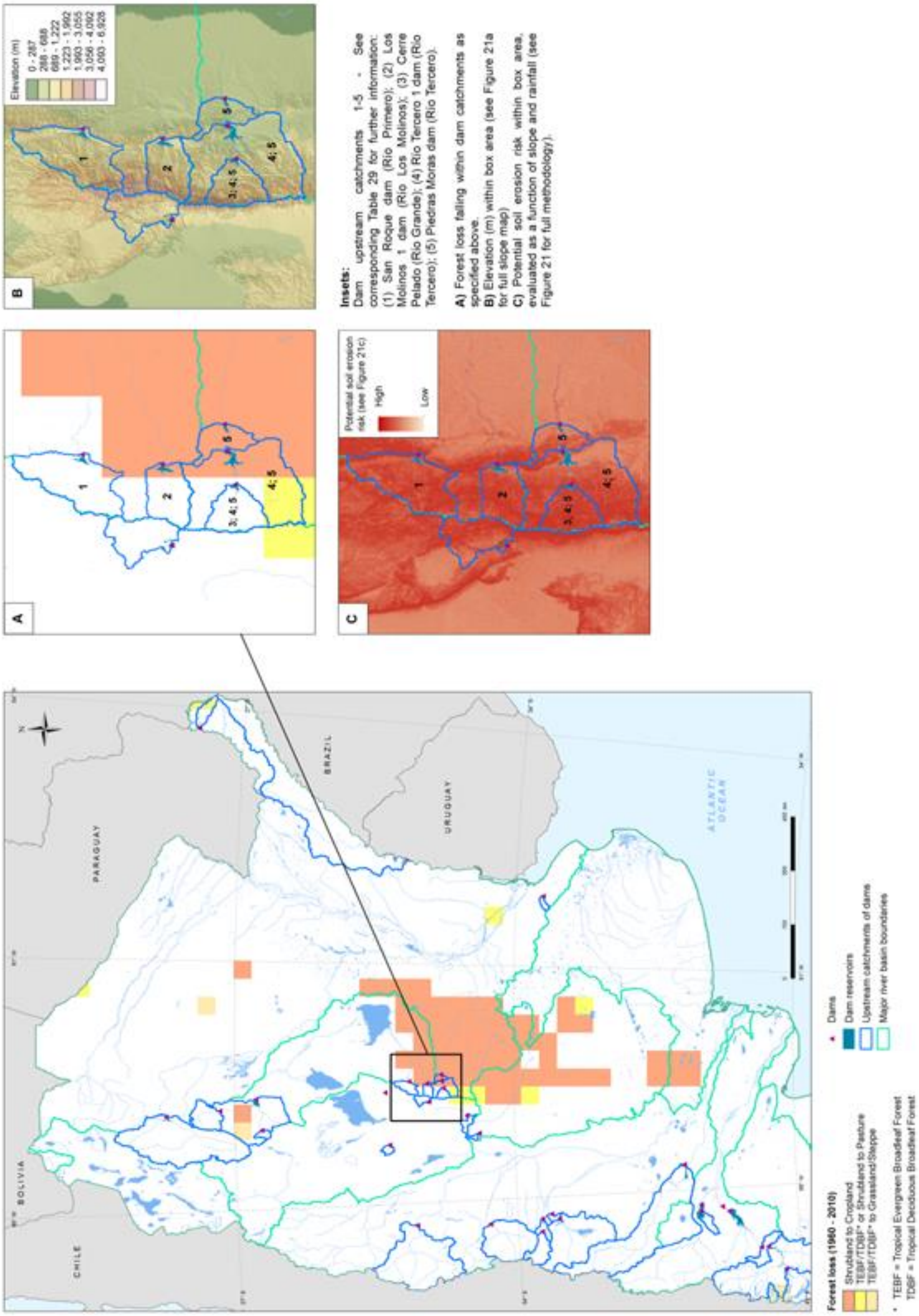


Figure 21: Identification of soil erosion risk and impact of deforestation on dam catchments

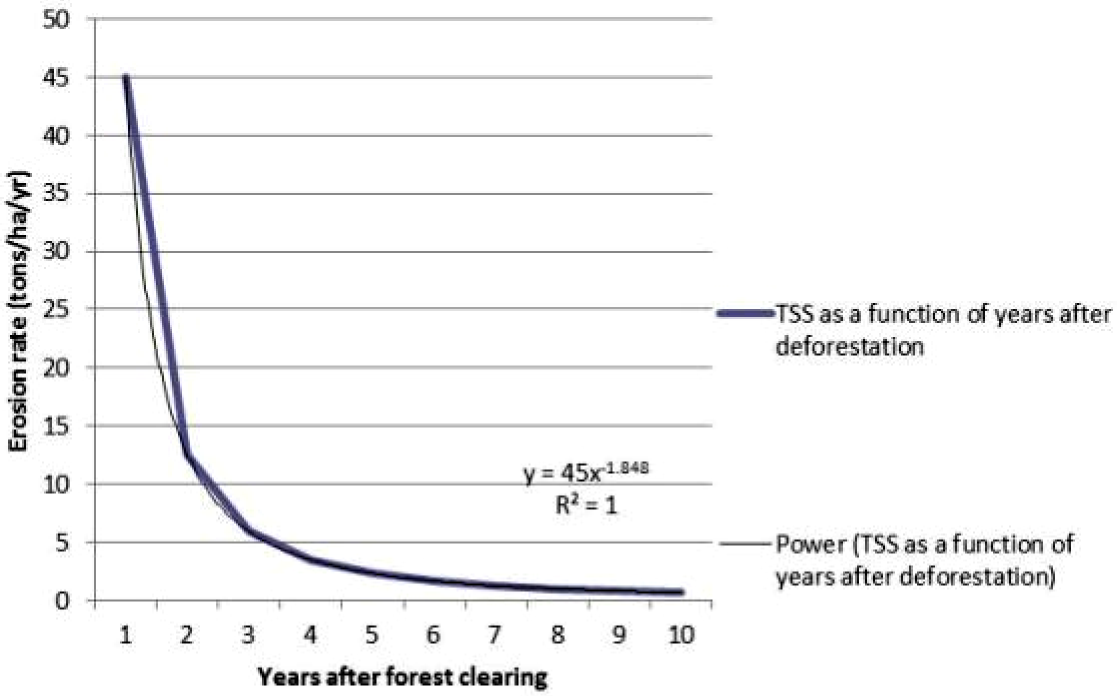


Figure 22: The effects of deforestation on erosion and siltation over time
Source: Crafford et al., 2012

Table 27: Quantity of soil (t) eroded in dam reservoir catchments as a result of deforestation

	1960-70	1970-80	1980-90	1990-2000	2000-2010
Río Tercero 1	13,102,200	0	18,374,400	12,075,600	0
San Roque	991,800	0	0	0	0
Los Molinos 1	2,992,800	0	0	0	0
Río Hondo	43,500,000	40,890,000	0	0	0
Cabbra Corral	0	2,610,000	0	0	0
Piedras Moras	0	0	5,272,200	0	0
Alicura	0	0	14,894,400	0	0
Cerre Pelado	0	0	0	295,800	0
Uruguay	0	0	0	0	15,312,000
Total	60,586,800	43,500,000	38,541,000	12,371,400	15,312,000

Applying the value of 74 tons of soil loss per hectare over a decade to the area deforested, excluding any areas with low risk of erosion (Figure 20), gave the amount of soil eroded in dam catchments (Table 27). Assuming that all of the additional sediment load would have been deposited in downstream dam reservoirs[28] (where they exist), at a sediment density of 1.95 tons/m^3 (Crafford et al., 2012), the cumulative loss in water-storage capacity due to deforestation exceeds 87.3 million m^3 by 2010 (Table 28).

[28] The travel time for all the eroded sediments to reach a reservoir will vary, ranging from years to centuries.

Table 28: Estimated cumulative lost water storage

	1960-70	1960-80	1960-90	1960-2000	1960-2010
Lost storage m3	31,070,154	53,377,846	73,142,462	79,486,769	87,339,077

Table 29: Hydro-electric dams in Argentina potentially affected by deforestation

Hydro-electric dam	Río Tercero 1	San Roque	Los Molinos 1	Rio Hondo
Year of dam construction	1936	1944	1953	1967
River sited upon	Tercero	Primero	Los Molinos	Dulce
Potency	600 MW	25 MW	54 MW	17 MW
Capacity generation	1,839 GWh (Est.)	77 GWh (Est.)	165 GWh (Est.)	99 GWh
Reservoir capacity (million m3)	560	201	307	1740

Contd.	Cabbra Corral	Piedras Moras	Alicura	Cerre Pelado	Salto Grande
	1973	1979	1984	1984	1991
	Juramento	Tercero	Limay	Grande	Uruguay
	102 MW	6.3 MW	1,050 MW	750 MW	1,890 MW
	250 GWh	19 GWh (Est.)	2,360 GWh	970 GWh	6,640 GWh
	3100	90	3215	370	2750

The costs in terms of lost potential hydropower production were estimated for each dam. Potential lost electricity production was based on the lost water storage capacity for each dam, which was calculated using the erosion data and assuming that the efficiency of water use (which also accounts for evaporation from dams) is 0.36 GWh per m^3 on average across the hydro-electric power stations. This is a conservative estimate and an efficiency of water use of double this value might be possible (this is considered below). The value of electricity used was the producer price of US$7.80 MWh (with an increase from 2013, reflecting the government policy to increase the fixed rate (Pampa Energia, 2014)). This may underestimate the actual value of electricity to the economy of Argentina.

Table 30: Cost of lost hydro-electricity (2010 US$) from year of loss up to 2099 resulting from the deforestation occurring in each decade with 1% and 5% discount rates

Decade of loss	5%	1%
1960-70	6,455,609	10,705,429
1970-80	4,008,586	7,059,864
1980-90	2,996,617	5,700,049
1990-2000	783,746	1,651,529
2000-2010	749,544	1,823,594

With a higher rate of efficiency of water use of 0.72 GWh per m^3 the above values would be doubled. However, since this difference would result in a change in the order of tens of millions of dollars, compared with the value of land-use change gains measured in hundreds of billions of dollars, it is not considered important to further increase the accuracy of these estimates. Nevertheless, in other countries the impact of soil erosion on hydro-electric production could be more significant. This could also apply to an analysis of the costs and benefits of deforestation at the scale of an individual water catchment in Argentina.

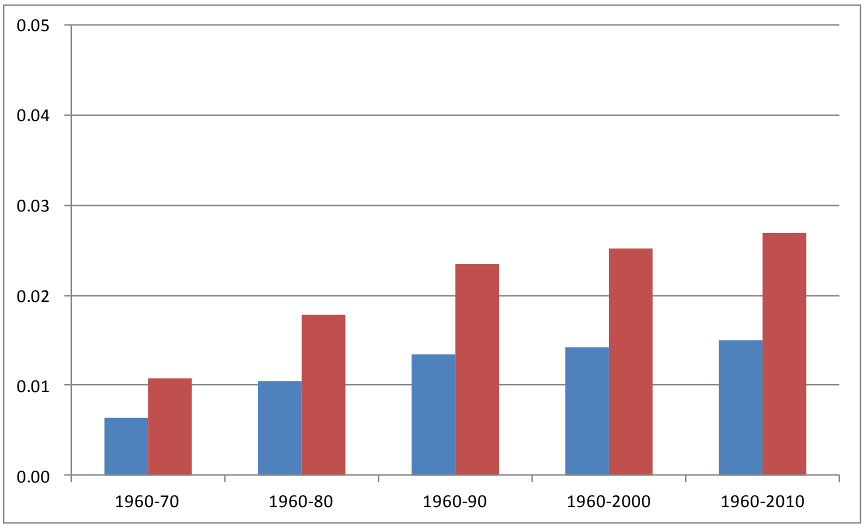


Figure 23: Cumulative potential hydro-electricity production losses to 2099 from deforestation each decade (2010 US$ billion), (5% discount rate blue bars, 1% discount rate red bars)

6.6.7 Impacts on water flows: floods and droughts

The deforestation impact on water flows was assessed for both floods and droughts. The area of deforestation each decade in main water basins in northern Argentina was assessed (Table 33) using the HYDE data (see Appendix B for more detail). Once the percentage area deforested is calculated, the additional flood risk can be estimated.

Table 31: Estimated area of deforestation each decade in main water basins

Name of major basin	Basin area (hectares)	1960-70	1970-80	1980-90	1990-2000	2000-2010
La Puna Region	8,458,700	0	0	0	0	0
La Plata	84,300,600	3,699,300	264,500	2,046,200	1,160,100	779,100
Mar Chiquita	12,967,100	1,436,700	235,500	453,200	0	250,000
Salinas Grandes	17,752,500	0	0	0	133,700	0
South America Colorado	37,477,100	0	0	0	0	0
Negro	16,294,100	0	0	181,500	0	0
South Chile	4,777,300	0	0	44,400	0	0
Pampas Region	17,556,100	487,200	152,600	1,083,100	1,817,700	500,000
North Argentina, South Atlantic coast	22,331,700	126,900	97,400	0	388,500	500,000
Total	**221,915,200**	**5,750,100**	**750,000**	**3,808,400**	**3,500,000**	**2,029,100**
Total cumulative		**5,750,100**	**6,500,100**	**10,308,500**	**13,808,500**	**15,837,600**

Estimating the costs of floods caused by deforestation is a complex task[29], and in order to make it a realizable one in this instance some general assumptions must be adopted. Previous research has

[29] Since a multitude of factors will influence flood risk in addition to changes in land cover from forests to crops or pasture, such as topography and geology.

found that a decrease in natural forest area of 10% results in a model-averaged prediction of flood frequency increasing between 4% and 28% (Bradshaw et al., 2007). Therefore, the midpoint figure of 4-28%, i.e. 16%, is used as the measure for change in frequency each decade if forest cover reduces by 10%[30]. However, the magnitude of the response has been questioned by some (Van Dijk et al., 2009), thus the impact of this assumption is considered in section 6.7.

This change in frequency in the Argentinean context is equal to 1.12 additional floods per decade. With total damage of US$2.8 billion (1990-2000), and US$400 million per flood on average for Argentina as a whole (data from the OFDA/CRED International Disaster Database quoted in Bradshaw et al., 2007), this means that a 10% change in forest cover could incur damage of US$448 million. A linear damage relationship is assumed, so that a 1% loss results in US$44.8 million damages (per decade), i.e. there is no threshold effect (a simplifying assumption). Applying the percentage area deforested to the total catchment area, means that from 1960 - 2010, deforestation caused flood damage losses of a little over US$1 billion[31] (in those decades).

The flood risk will continue into the future, so following the convention of looking forward up to year 2099 and discounting at 1% and 5%, the historic plus the discounted values to 2099 are as in Table 32.

Table 32: Cost of flood damage (2010 US$ billions) from year of loss up to 2099 resulting from the deforestation occurring in each decade with 1% and 5% discount rates

Decade of loss	5%	1%
1960-70	8.10	12.62
1970-80	0.90	1.50
1980-90	3.82	6.82
1990-2000	2.81	5.56
2000-2010	1.22	2.82

With regard to low flow, it is assumed that Argentine forests can increase dry season river flow, as found in Panama by Ogden et al. (2013) (see section 6.5 for further discussion). In the dry season in Kenya (five months from June to October) around 2,000 m^3 of additional water per hectare can flow from mountain forests compared with cleared land (Crafford et al., 2012). On average Argentina receives only 21.7% of the precipitation occurring in the Kenyan mountain forests, so it is unlikely that the average hectare of forest in Argentina would be responsible for the same volume of water flow[32]. Based on the lower rainfall, we calculated that dry season flow would be 20% of the Kenyan amount,

[30] There are also indications that a 10% increase in forest cover can reduce the number of households evacuated by 16% (Tan-Soo, 2010).
[31] This can be compared with recorded actual flood damages: just the top five floods between 1983 and 2013 caused almost US$6 billion of losses.
[32] The difference in water flow following deforestation will depend on the subsequent land use - Oyarzún et al. (2004-5) found that native forests in Chile yielded 10,000 m^3 of additional water flow per hectare compared with a catchment deforested and replaced with plantation forests.

i.e. 400 m^3 per hectare over five months[33]. To refine this simple analysis, topography and geology could be considered; with data ideally being used from the location being analyzed or areas with similar characteristics, where this is possible, in order to determine whether there is likely to be a net increase or decrease in dry season flow as a result of deforestation in the area.

Table 33: Reduction in dry season water flow as a result of deforestation

	1960-70	1970-80	1980-90	1990-2000	2000-2010
Total basin area deforested (ha)	5,750,100	750,000	3,808,400	3,500,000	2,029,100
Water loss (billion m3)	23.0	3.0	15.2	14.0	8.1
Total cumulative basin area deforested (ha)	5,750,100	6,500,100	10,308,500	13,808,500	15,837,600
Cumulative water loss per dry season (billion m3)	23.0	26.0	41.2	55.2	63.4

The above analysis only estimates river-water flow through forest areas. Water vapour needs to rise and condense in order to create clouds and then rain, and forests can aid this process, whilst croplands can deepen droughts (Oglesbuy et al., 2010). Over the dry season on average, precipitation is 25 mm per month, and evapotranspiration from forests in South America can be 50% of the precipitation (Vourlitis et al., 2002). However, deforested land used for crops and pasture will also produce evapotranspiration. Cropland only does so at half the rate of forests (Evans, 2012), resulting in a net loss of evapotranspiration of 20%. Of this, 57% is subsequently returned as precipitation to the land (van der Ent et al., 2010), which given prevailing winds is assumed to be in Argentina. This equates to 8.55 mm (or 85.5m^3 per hectare) over each dry season.

Table 34: Reduced precipitation across the region due to lost evapotranspiration

	1960-70	1970-80	1980-90	1990-2000	2000-2010
Total basin area deforested (ha)	**5,750,100**	**750,000**	**3,808,400**	**3,500,000**	**2,029,100**
Water loss (billion m3)	**4.9**	**0.6**	**3.3**	**3.0**	**1.7**
Total cumulative basin area deforested (ha)	**5,750,100**	**6,500,100**	**10,308,500**	**13,808,500**	**15,837,600**
Cumulative water loss (billion m3)	**4.9**	**5.6**	**8.8**	**11.8**	**13.5**

Table 35: Combined effects of reduced water flow and reduced precipitation

	1960-70	1970-80	1980-90	1990-2000	2000-2010
Total basin area deforested (ha)	**5,750,100**	**750,000**	**3,808,400**	**3,500,000**	**2,029,100**
Water loss (billion m3)	**27.9**	**3.6**	**18.5**	**17.0**	**9.9**
Total cumulative basin area deforested (ha)	**5,750,100**	**6,500,100**	**10,308,500**	**13,808,500**	**15,837,600**
Cumulative water loss (billion m3)	**27.9**	**31.6**	**50.0**	**67.0**	**76.9**

Obviously there are many uncertainties associated with these estimates, such as the quantity of dry season flow (which in some dry forests may even be negative (Bosch and Hewlett, 1982)), and assessment of where evapotranspiration is eventually returned to the land or sea as precipitation. Thus a more detailed analysis by forest type, taking into account soils and local rainfall patterns, would

[33] On average the Chaco region of Argentina receives 7,600 m^3 per hectare of precipitation each year, but in the dry season the total amounts to just 500 m^3 per hectare.

yield more accurate water flow estimates. For instance, the northern tropical forests in Argentina may play a greater role in reducing droughts than the dry forests further south.

One way to estimate the impact and costs of droughts is to multiply the lost water volume by the mean tariff for water and sanitation (US$0.79 per m^3 (Pan American Health Organization, 2000). Assigning 50% of this value to each service gives a water supply value of US$0.395 per m^3. However, where groundwater abstraction is a possibility, the cost of alternative water supplies would be much less than this (less than US$0.02 per m^3 if just costing a pump and fuel (Wichelns, 2010)). In some areas, a value closer to this may be more realistic (though costs will vary in terms of local fuel prices, as well as drilling a borehole which will depend on cost of machinery hire, geology and water table depth). However, how sustainable over the long-term this use of groundwater would be depends on extraction and replenishment rates. In the current analysis, an average value of US$0.04 per m^3 is used[34]. The impact of this assumption is explored in section 6.7, as adopting a much higher value could be valid, and makes a substantial difference to the result of the analysis as a whole.

Table 36: Estimated cost of drought (2010 US$ billions) from year of loss up to 2099 resulting from the deforestation occurring in each decade with 1% and 5% discount rates

Decade of loss	5%	1%
1960-70	7.79	12.14
1970-80	0.87	1.44
1980-90	3.68	6.56
1990-2000	2.70	5.35
2000-2010	1.17	2.71

[34] The marginal physical product of stream water in the summer season in Chile was estimated to be over twice this value (Nunez et al., 2006), at US 0.09 per m^3 (when adjusted to current prices).

To estimate the value of water flow services lost to deforestation, the impact of both floods and droughts are summed (Figure 24).

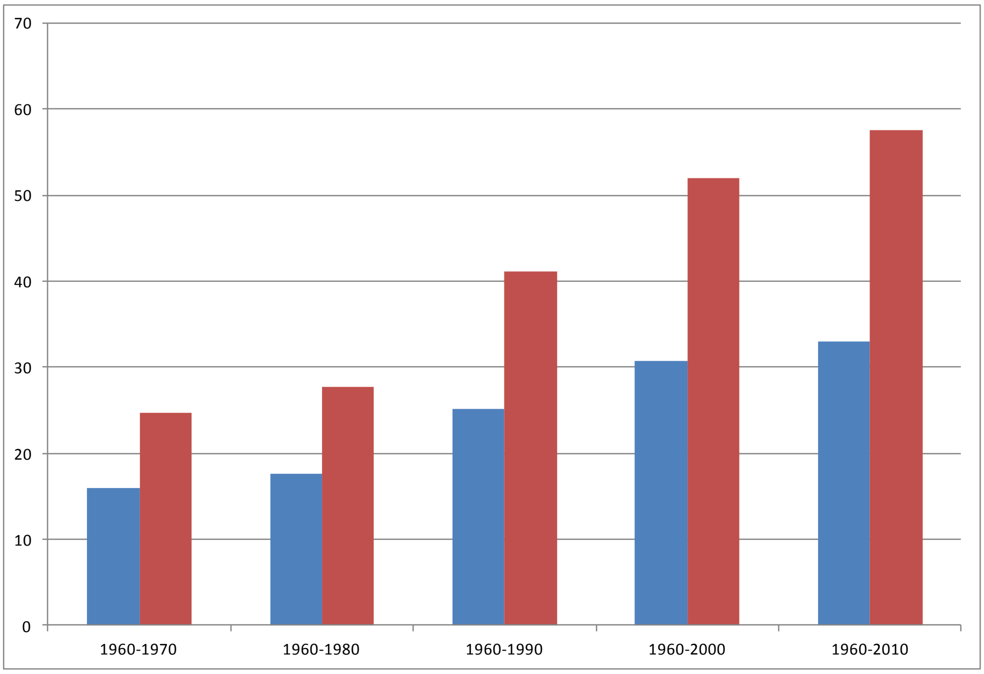


Figure 24: Water flow regulation, i.e. combined impacts of flood and drought, cumulative impacts (2010 US$ billion) up to 2099, with discount rate of 1% (red) and 5% (blue)

These estimates could be on the conservative side, since a growing population in Argentina over the remainder of this century may lead to a greater demand for water (and the cost of groundwater, as well as the value of water could increase), whilst expansion of urban areas could increase the number of properties and infrastructure at flood risk over coming decades (and growing household wealth increases the magnitude of potential losses from flooding events). However, the biggest influence on water flow regulation values will be the underlying physical assumptions, which can only be improved by undertaking much more detailed hydrological modelling. Since the water regulation values are relatively large for Argentina, this is an area that is worth further investigation. Refinement of the estimated values, including the underlying hydrological assumptions[35], should be considered a priority in any further iteration of this analysis.

6.6.8 Biodiversity and landscapes as a tourism resource

To assess the lost tourism resource, the areas of highest tourist potential were identified based on both Protected Area designation and significance for biodiversity (Figure 25). These were considered relevant proxies; however, more specific information on tourists' preferences could be used alongside more detailed spatial data on land cover, topography, and other relevant factors, to pinpoint potential tourist sites. The approach adopted here means that an average per hectare monetary value will have to be used when the economic valuation is undertaken.

[35] Especially the impact of the Chaco forests on dry-season flow.

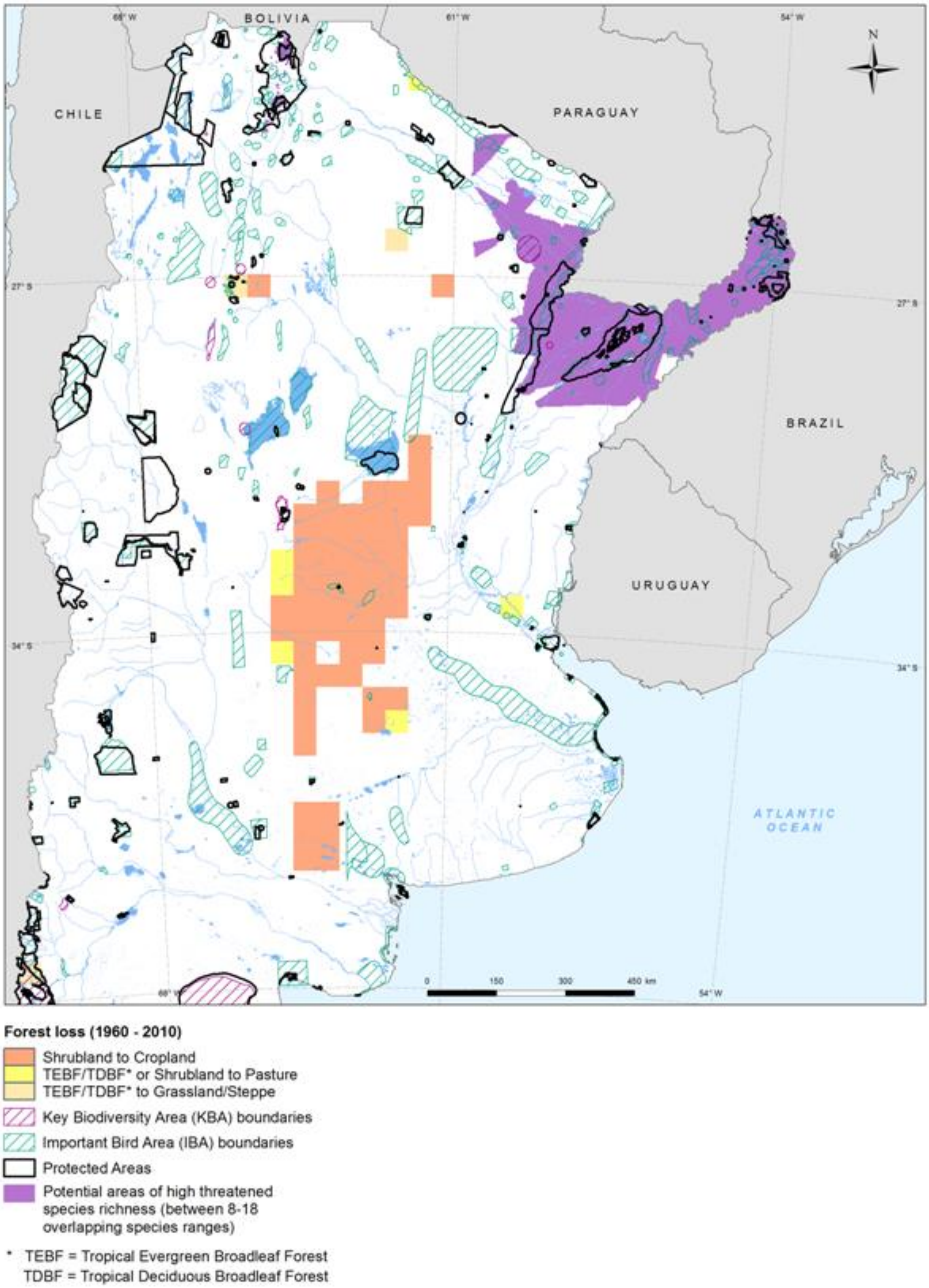


Figure 25: Forest loss and areas of potential importance for tourism

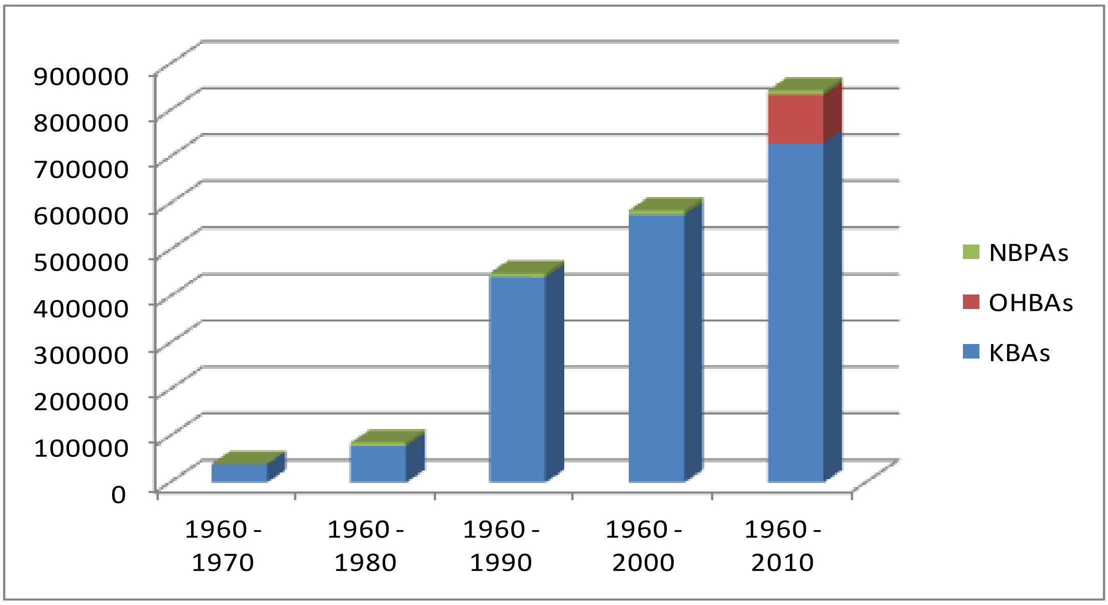


Figure 26: Cumulative area (ha) of forest lost that could have had a tourism value based on high biodiversity presence or designation. NBPAs = Protected Areas; KBAs = Key Biodiversity Areas; OHBAs = Other High Biodiversity Areas

Using figures for the value of forests for ecotourism in Ecuador (Verweij et al., 2009) of US$6.99/ha (in 2010 dollar value), and applying a reduction of 4% a year projecting backwards to take into account the growth rate of international tourism to Argentina over the last 50 years (Kester, 2009), an average decadal value is applied to the number of lost forest hectares in Protected Areas, Key Biodiversity Areas, and Other High Biodiversity Areas. This ignores areas that have been lost but might now be assessed as valuable for biodiversity. Long-term tourism growth forecasts tend to run to no further than 2030, and generally reflect past growth trends. However, with drastic CO_2 emission reductions required by the second half of this century coupled with resource limitations, the cost of transportation is expected to rise. Whilst there is little research about the implications of CO_2 reductions for tourism there has been greater consideration of the impact of peak oil production on tourism, finding that growth would be reversed (Becken & Lennex, 2012; Logar & van Der Bergh, 2013). Therefore, the forward value to 2099 is based on a more conservative (compared with UNWTO) annual increase of 2%, though this may still be optimistic.

However, domestic tourists can also be considered. As developing countries reach upper middle income status, their populations are willing to pay increasing amounts toward forest conservation. Studies in high income countries have found that populations highly value forests as a tourism and recreation resource, and this importance grows as countries shift from middle to high income, yet government spending on conservation programmes lags far behind (Vincent et al., 2014). There are likely to be various reasons for this, but sometimes governments may simply be unaware of what their public values. For example, a case-study from Malaysia (Vincent et al., 2014) found that while the local government is reluctant to close an area of forest to illegal logging for fear of economic loss, the public weighs the value of conservation for society as a whole above the economic security of a minority of loggers. However, as there is no data available to assess the situation in Argentina, the value of forests for national recreation and tourism is not included in the analysis. It can at least be noted that overall domestic tourism is significant, with 48 million tourists in 2012, almost 10 times the number of international visitors (Vertical Edge, 2013).

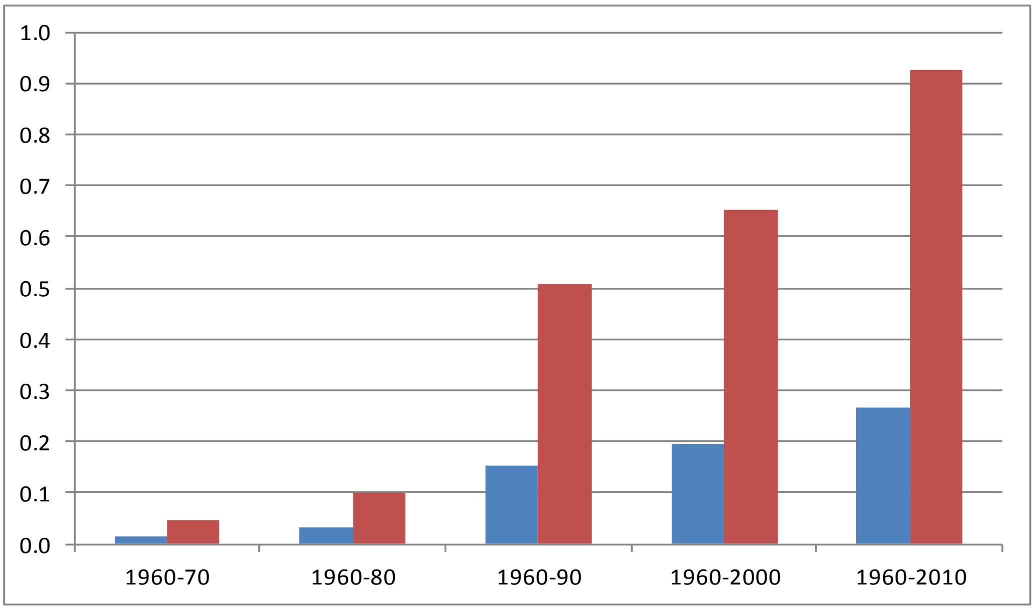


Figure 27: The cumulative cost to 2099 of lost tourism income (2010 US$ billion) (5% discount rate blue bars, 1% discount rate red bars)

Other cultural values are also excluded in this analysis. Cultural values are difficult to place a monetary value on. They can be particularly important for indigenous peoples. Therefore, alternative approaches should be used to assess their importance. But in an economic (i.e. cost:benefit) evaluation framework monetary valuation is required. So instead, these values should be noted, rather than ignored altogether, and highlighted as additional considerations for decision-makers when they are evaluating options that impact on cultural values.

As a footnote to the analyses in this section, it should be emphasized that the estimates produced for lost ecosystem services could be improved if more time and resources were available to undertake the task. Better historical land-use cover maps, a more detailed analysis by forest type, and collecting primary data in the field could increase the accuracy of estimates. However, at the national scale, the findings could broadly reflect the magnitude of ecosystem service losses. In addition, some data refinements require a better understanding of basic ecological and physical processes, which necessitate large-scale experiments over a period of years. In the meantime it is preferable to attempt an analysis with the current knowledge-base, but being aware of the many uncertainties involved.

6.7 Estimation of corrected economic gains from past deforestation

The gains from deforestation, namely the wood products from cleared land plus the past and future agricultural (soy and beef) production are summed (i.e. those reported in Figure 3-5). These are the present values (using the 1% and 5% discount rates) from the time of deforestation up to 2099 for each period. The costs resulting from deforestation are also summed (as calculated in section 6.6); noting that these are an incomplete sub-set of the total lost ecosystem services.

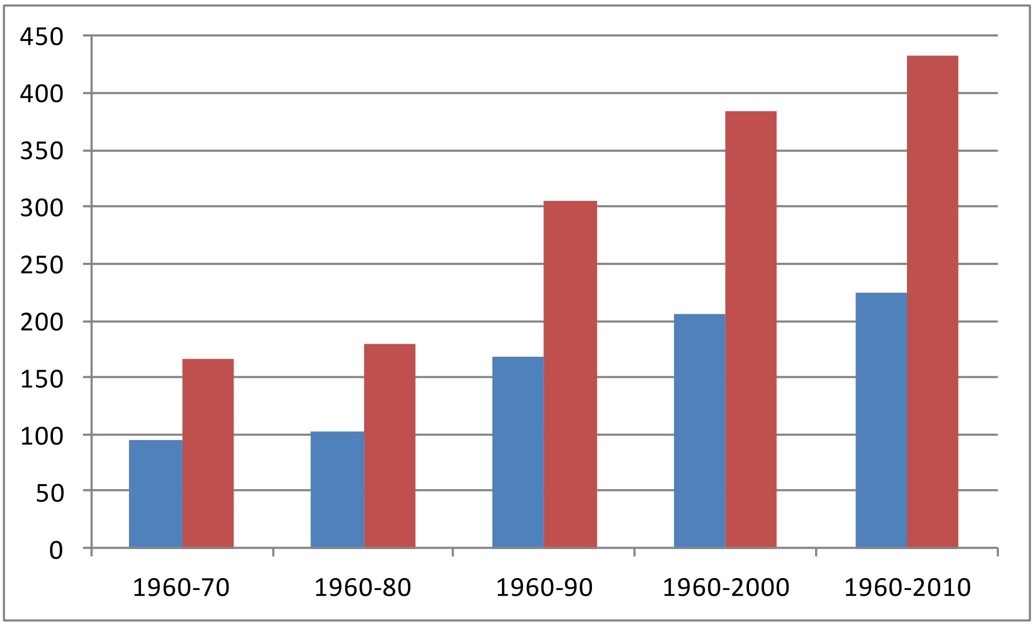


Figure 28: Cumulative gains to 2099 from deforestation over the period 1960-2010, present value (2010 US$ billion) using 1% discount rate (red) and 5% (blue)

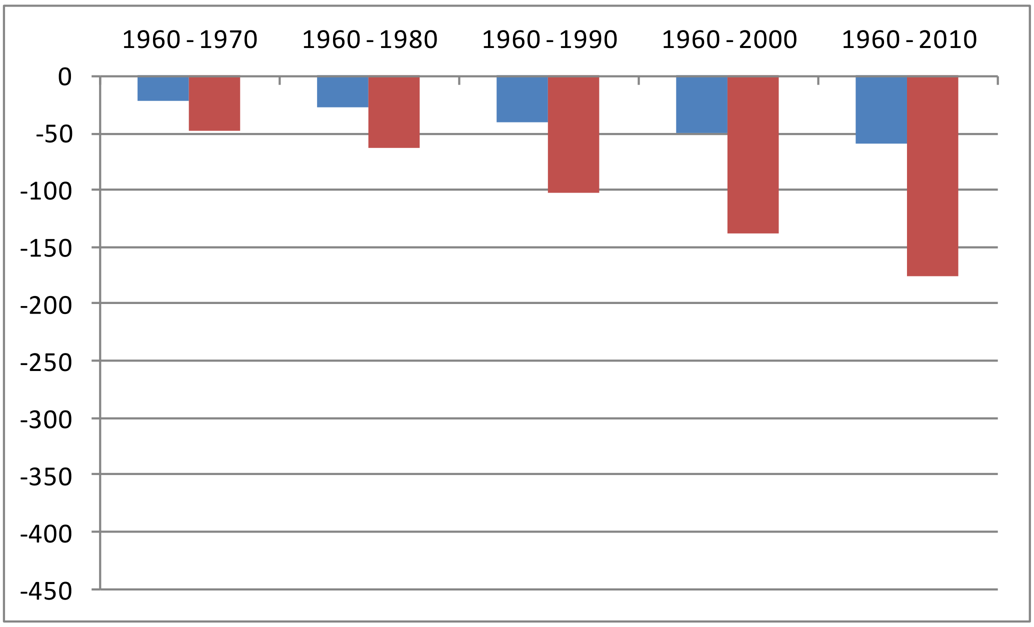


Figure 29: Cumulative losses to 2099 from deforestation over the period 1960-2010, present value (2010 US$ billion) using 1% discount rate (red) and 5% (blue)

Whilst the above graphs show an overall gain from past deforestation between 1960 and 2010, the magnitude of this is significantly lower than the uncorrected figures would indicate: with a 5% discount rate the unaccounted costs of forest ecosystem service losses are around a quarter of the benefits from forest conversion, whilst with a 1% discount rate these costs are over 40% of the benefits. The ongoing environmental impacts resulting from deforestation are ignored to a much greater extent when a higher discount rate is used (although future agricultural production is also valued less with a high discount rate). If sustained national economic success over the long term is an objective, then economic planning decisions should consider using a low discount rate when they impinge on ecosystem functioning.

Some of the assumptions adopted in the analysis are very influential on the final results. The most sensitive estimates are related to water flow regulation and to sustainable wood products harvesting (to a much lesser extent). For instance, if the water flow regulation benefits only occur where there is tropical forest cover, then the overall losses reduce from US$177 billion to US$126 billion (1% discount rate). However, if future timber prices increase by their historical rate this adds an additional US$12 billion to the 1960-2010 cost (1% discount rate), and if flood frequency increases by the estimate at the top end of the range this adds a further US$22 billion (at the lower end of the range it reduces by the same amount).

The value of water is the most influential variable. If the higher marginal price of water equivalent to the service charge for water supply is used (which could be justified if: (a) droughts are widespread and impact on households and businesses which do not have access to private groundwater supplies, and (b) the service charge is considered representative of the marginal value of water in Argentina), then the additional costs for 1960-2010 increase by US$250 billion (assuming that all the water is used). Just this latter adjustment results in a total costs estimate which negates all of the benefits from deforestation between 1960 and 2010.

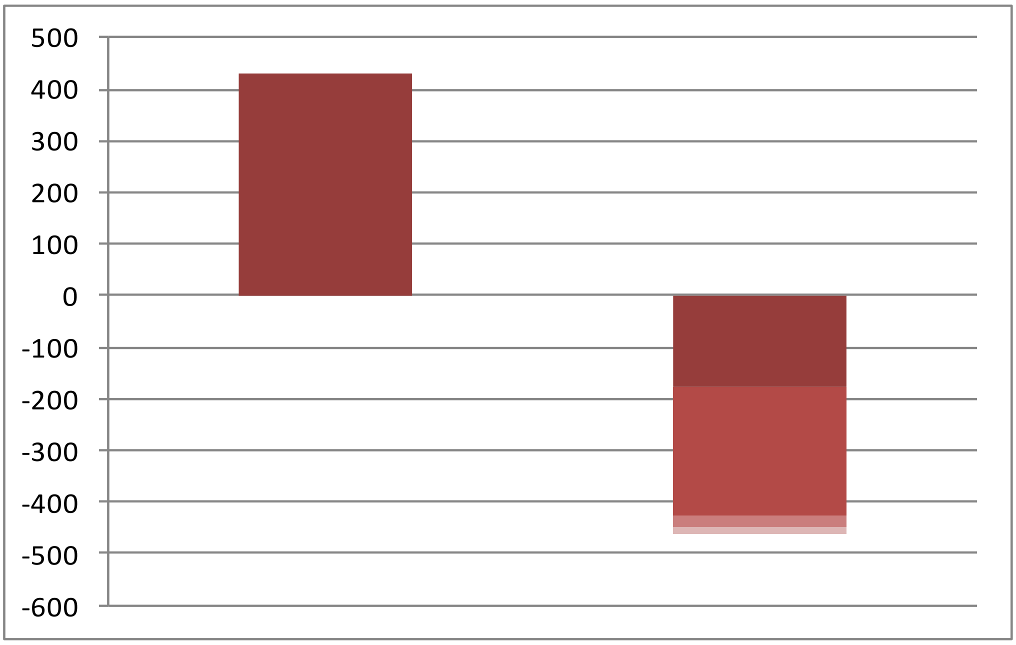


Figure 30: The benefits (left) and costs (right, with adjusted drought (light pink), flood (pink) and wood (light red) values)) from deforestation 1960 – 2010, up to 2099 using a 1% discount rate (2010 US$ billion).

See Appendix C for a table of the results, presenting the full range of values reflecting the assumptions discussed above. This range, from net gain to net loss, reflects the uncertainty around estimating the values associated with forest ecosystem services[36]. Obviously even a net loss does not mean that no forests should have been converted to agricultural land in Argentina. Conversion in earlier decades could have been beneficial. However, the extent of recent changes may be less so.

Taking the period, 1960–2010, as a whole obscures some of the marginal changes in costs and benefits between decades. For instance, in the period 2000-2010 the additional agricultural and wood products gains associated with the additional deforestation that decade are valued at US$46 billion (using the 1% discount rate). However, the costs were also high (partly reflecting the increasing cost of carbon), so that the overall net gain was just US$7 billion (up to the year 2099) or could even be a large net loss of US$17 billion if the higher value of water is used.

Conversion of land will be much more beneficial in some locations than others. Locations where forests contain high carbon stocks, and/or are important for hydrological functions, and/or could deliver valuable sustainable harvests of wood and NWFPs, are likely to have a high existing value, which may be greater than the agricultural value if converted. Detailed spatial planning can help identify these high value forests, and can also include other important factors such as cultural and biodiversity value.

Forest ecosystem values not covered by this analysis include pollination, genetic value, and cultural values beyond international tourism (for instance, in addition to recreation values, there are national and global non-use values associated with important habitats and species, demonstrated through donations to conservation charities). Based on the global estimates of pollination benefits (Stathers, 2014), these could add an additional US$5-10 billion to the costs of deforestation in Argentina between 1960 and 2010.

The considerable uncertainty of the future value of both these non-estimated and estimated forest ecosystem services should ideally also be accounted for in an economic analysis of deforestation. This is particularly relevant in the context of native or old-growth forests, for which deforestation is an irreversible decision, as the quality of future forest regrowth (in terms of ecosystem service provision and biodiversity) is likely to be much reduced. The lost potential for such forests to harbour threatened species valued in their own right, but also as a potential source of genetic material for new medicines and crop varieties, may be especially important. Accordingly, there is significant option value in

[36] There are few other studies that assess the monetary value of multiple forest ecosystem services at a national level, especially at more than one point in time. However, Gundimeda and Atkinson (in UNU-IHDP and UNEP, 2014) undertake such an estimation using a top-down methodology (as opposed to the map-based approach adopted here) for various countries, including Argentina. They find that between 1990 and 2010 the value of Argentina's forest that was lost (based on a similar range of ecosystems services as identified in section 6.5, above) was almost US$445 billion (in 2010 dollar values). This compares with less than US$75 billion in Figure 29. The main differences are that additional supporting and regulating services are included, such as biological diversity (totalling almost US$100 billion), as well as recreation values beyond just international tourism (around US$140 billion). However, as Gundimeda and Atkinson note, their analysis is preliminary and generic, and a bottom-up spatial approach to valuation will be more accurate.

delaying the decision to clear areas of native or old growth forests until the full value of all the ecosystem services they can provide in future years becomes more certain (Forsyth, 2000).

A further consideration is the distribution of the costs and benefits associated with the deforestation. In general, the benefits of soy and beef production accrue to the producers, with farm workers and those in the supply chain (both upwards and downwards, e.g. processors and input suppliers) also gaining (as well as consumers, mainly in other countries). As 6% of Argentine farmland is under foreign ownership, the profits from this production will most likely be sent abroad. The losers are those who are no longer able to make use of the forest for NWFPs and sustainable wood products, as well as those impacted by increased flooding, droughts and climate change impacts (much of which will be current and future generations over the remainder of this century).

However, expansion of agro-exports based on GM soy mono-cropping has been promoted as a national economic development model for the last two decades. Since 2003, export taxes have become an important source of total government revenue, and in 2011 the government revenue from soy exports taxes were around US$8 billion dollars (Leguizamón, 2014). Government assistance in the form of social funds have been distributed to those people who lost access to the forest as a means of subsistence, and many small rural towns now rely on soy-derived money as a significant part of their budget (Leguizamón, 2014). Therefore, some of those who are losers have been compensated (at least partly). The remaining costs are to future generations, who in theory could gain to some extent if investments have been made in national infrastructure between 1960 and 2010 that improved the long-term economic prospects of Argentina. If this is not the case, then the deforestation over this period simply becomes a matter of wealth redistribution from future generations to fund current consumption (with most gains accruing to a minority of the population).

It could be the case that deforestation provides the means to spur economic development in Argentina. To briefly evaluate the effectiveness of deforestation for economic development, Argentina is compared with fellow South American countries, two of which have increasing forest cover and one of which has held forest cover fairly constant. Together they had a little less forest cover than Argentina at the turn of this century (Table 37).

Table 37: Forest area in Argentina 1990 – 2010 compared with other countries in the region (FAO, 2015)

	Forest area (1 000 ha)		
Country	**1990**	**2000**	**2010**
Chile	15 263	15 834	16 231
Suriname	15 430	15 931	15 351
Uruguay	798	1 370	1 731
3 combined	31 491	31 765	33 313
Argentina	34 793	31 860	28 596

In 1990, Argentina had a greater forest cover than the three other countries combined, but by 2010 had fallen below these. Therefore, for the 1990 – 2010 period as a whole, Argentina saw declining forest cover as a result of deforestation for conversion to agricultural land, whereas the three other countries combined experienced a trend of increasing forest cover. In Chile, this increased forest area involves plantations replacing natural forests, which will change the quantity of ecosystem services delivered. For the 1980 - 1990 decade as a whole (i.e. before the agro-export development model took off) Argentina had a higher Gross National Income (GNI) per capita compared to the average of the other three countries. However, from 2000 onwards the situation was reversed. Examples exist elsewhere in the continent of how high economic growth can be achieved whilst maintaining or increasing forest cover. Clearly, this is not to say that deforestation played any role in the lower economic performance of Argentina: countries face multiple economic challenges which can derail development plans. However, it is possible that alternative approaches may have been able to deliver an overall improvement in the Argentine economy, without the need to resort to deforestation and the loss of valuable ecosystem services.

With regard to future economic planning, it should be noted that value of future deforestation is not the same as the in past, because climate impact costs are increasing, so deforestation in 2010 - 2020, and each subsequent decade will be increasingly costly. There may also be growing economic impacts associated with other forest ecosystem service losses due to non-linear impacts on ecological functioning (passing ‘tipping-points’). Therefore, although the analysis presented here indicates that the deforestation that took place between 1960 and 2010 in Argentina may have delivered an overall economic gain (though the cost estimate may have exceeded the benefits if additional ecosystem services were included or water pricing assumptions were less conservative); future deforestation over the next 50 years may not show any net benefit. Therefore, a different approach to economic development, such as a Green Economy approach, might be more beneficial to the country over coming decades.

7. The economic drivers of deforestation globally

A brief analysis of the history of deforestation is useful to demonstrate how population growth and economic development have been drivers for deforestation, as well as placing current deforestation in developing countries in context. Deforestation has been seen as a prerequisite for economic development (at least in the early stages). Forests represent a source both of land and of wood (for fuel and timber), which is not always exploited sustainably. Hence the net loss of forests is unsurprising as the population grows.

It is likely that humans had an influence on both tropical and temperate forests as far back as 12000 BCE, by causing the extinction of medium and large vertebrates (Malhi et al., 2014) (Bonnicksen, 2000). The first evidence of human-caused deforestation appears in the Mesolithic period between 8400–7000 BCE (Brown, 1997), with larger areas subsequently being deforested as the keeping of livestock spread across Europe (Iyyer, 2009). The first documented records of deforestation come from China, which has detailed land-use records going back 3000 years. China's Loess Plateau was cleared of forest 3000-2500 years ago as logging took place and agriculture expanded due to a growing population (Zhang, 2000). Since then this land has been eroding and providing the sediment that gives the Yellow River its colour and name. As a result of the deteriorating environment the economic and cultural centres of China shifted eastwards. Despite early attempts at forest conservation (Thirgood, 1981), deforestation was widespread under the Roman Empire (Grull, 2012), possibly being a factor in the fall of Rome[37] (O'Sullivan et al., 2008), and in ancient Greece (van Andel and Demitrack, 1990). Research indicates that the ancient Mesoamerican civilizations of the Mayans and Aztecs amplified droughts in the Yucatán and southern Mexico by clearing rainforests to make room for agriculture as population grew, eventually also contributing to their collapse (Oglesbuy et al., 2010).

Significant deforestation took place in Western Europe from 1100-1500 as a result of the expanding human population. By 1500 there was a shortage of wood for heating and cooking in parts of Europe, reflected in the price of fuel-wood (Green, 2006). This deforestation was interrupted by outbreaks of bubonic plague, which caused farm abandonment and forest re-growth in Europe (Ruddiman, 2003). In the following centuries this demand rebounded, and there was also growing demand for timber for merchant ship building, as well as naval ships to protect trade routes and defend claims to newfound territories (Melby, 2012). Deforestation expanded into other continents as European colonies were established and populations grew[38]. Some of the impacts of deforestation were felt soon after it had occurred. In 17th century New England, changes were felt by the colonists in terms of local climate (higher summer temperatures) and hydrological functioning (there was greater run-off and soil erosion, with streams becoming dry for much of the year) after deforestation occurred (Cronon, 1983). Siltation in the Mississippi River as a result of deforestation lead to severe flooding and abandoned of colonial settlements in the 19th century (Norris, 1997).

[37] The siltation of estuaries, as a result of deforestation, has been postulated as a cause of increases in the incidence of malaria in the city.
[38] The rate of forest loss between 1850 and 1920 in 30 US states is comparable to that in the Amazon over the last 70 years.

The global maps of forest cover between 1770 and 2010 (Figure 31 and 32) reflect the expansion of colonialism. Whilst deforestation has continued until the present time, in recent decades forest cover has increased in many developed nations. This has led to the promotion of the 'forest transition' hypothesis. Although a 'forest transition' is observable in a number of countries, the process by which this is happening, and how applicable it is at the global scale, is open to question. The historical evidence points towards population growth being the initial driver of deforestation in the immediate area, but the link is nuanced. As the marginal cost of trade (i.e. transport cost per unit of wood product) decreases due to improvements in transport technology and infrastructure, then the population 'hinterland' that acts as the source of demand expands. With the expansion of international trade infrastructure and low unit costs, this means that effectively it is now the global population level that matters (at least in all accessible regions).

However, an opposing force to population growth has been technological developments in the agricultural, manufacturing and energy sectors. This includes the 'green revolution' intensifying crop production (which means that less additional land is required to meet rising food demand); more efficient use of materials, such as wood fibres in fibreboards, as well as a shift from wood to plastics; and a decline in the use of wood as a fuel by switching to fossil fuels, hydro, nuclear, and solar power. Thus whilst the increase in global population from 2 to 3 billion was accompanied by a loss of 0.2 billion hectares of forest, the increase from 3 to 4 billion people saw a smaller loss of 0.1 billion hectares, and from 5 billion to 6 billion only 0.05 billion hectares of forest (UNEP, 2014).

Observed declining rates of deforestation in relation to growing income in some more developed nations has lead to the hypothesis of a forest transition curve. This is a specific example of the Environmental Kuznets Curve (EKC) which proposes an inverted U-shaped relationship between environmental degradation and economic development (Stern, 2004). That is, environmental harms such as pollution or deforestation increase as the economy develops and then at some point of GDP, or GDP per capita, the rate declines and environmental recovery may begin. If the EKC hypothesis were true, then economic growth would not be seen as a threat to the environment, in fact rapid economic growth should be encouraged (even at the expense of the environment), as it would be the means to eventual environmental improvement[39] (Stern, 2004). The sooner the GDP tipping point is reached the sooner the environment would recover.

Possible explanations for this EKC are that it reflects the progress of economic development, i.e. from clean agrarian economy to polluting industrial economy and then to clean service economy; as well as the tendency of people with higher income having a greater preference for good environmental quality (Dinda, 2004). Some air quality indicators related to local pollutants show evidence of an EKC (Dinda, 2004) and statistical data for individual countries (e.g. in Europe, as well as the USA, China, the Republic of Korea, and Viet Nam) seem to demonstrate a forest transition curve with increasing forest cover in recent decades. However, after numerous years of research, the overall empirical evidence remains equivocal (Chowdhury and Moran, 2012).

[39] Assuming there are no species extinctions, which would make full recovery impossible.

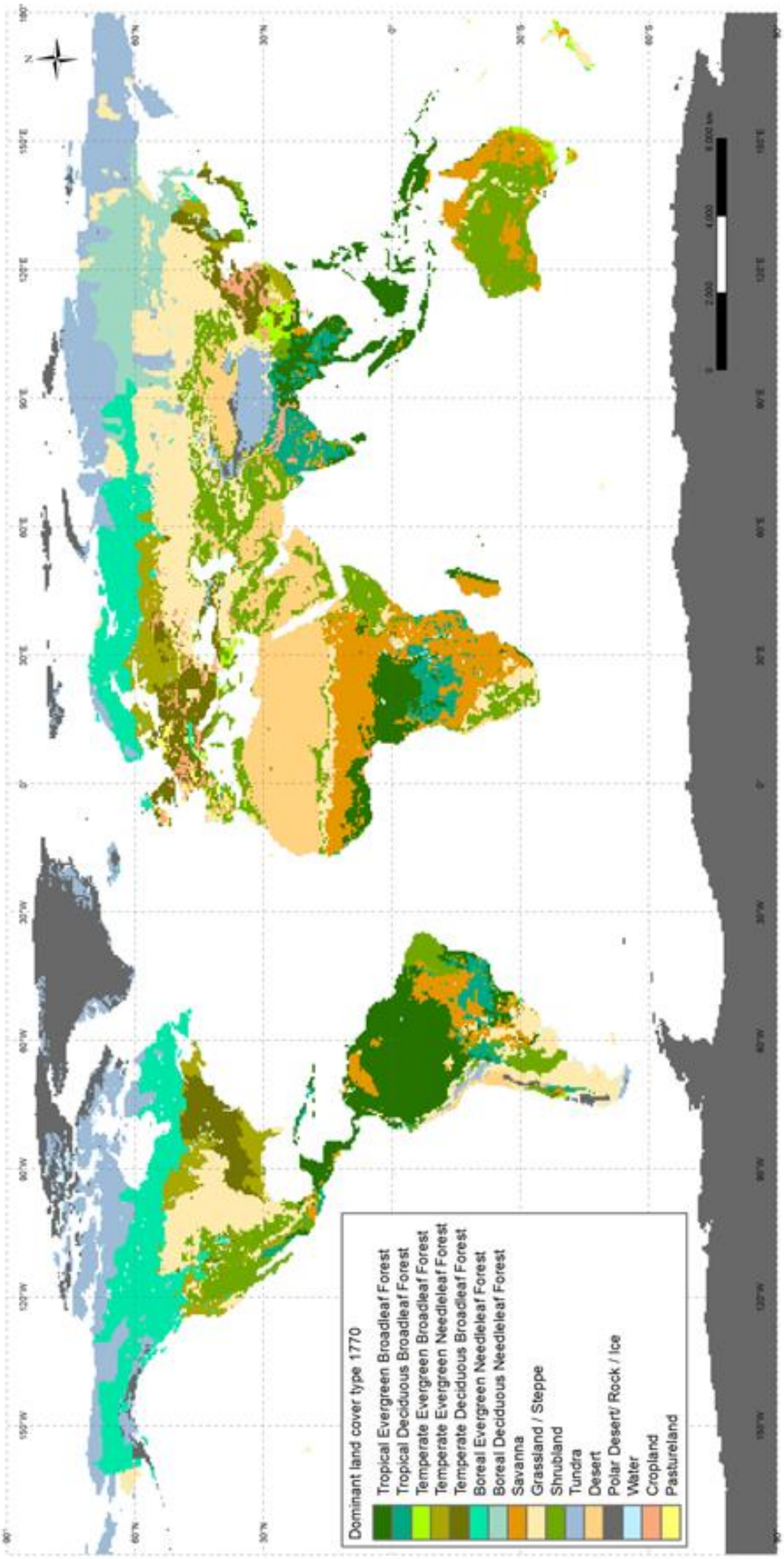


Figure 31: Global land cover in 1770 as estimated by the HYDE model

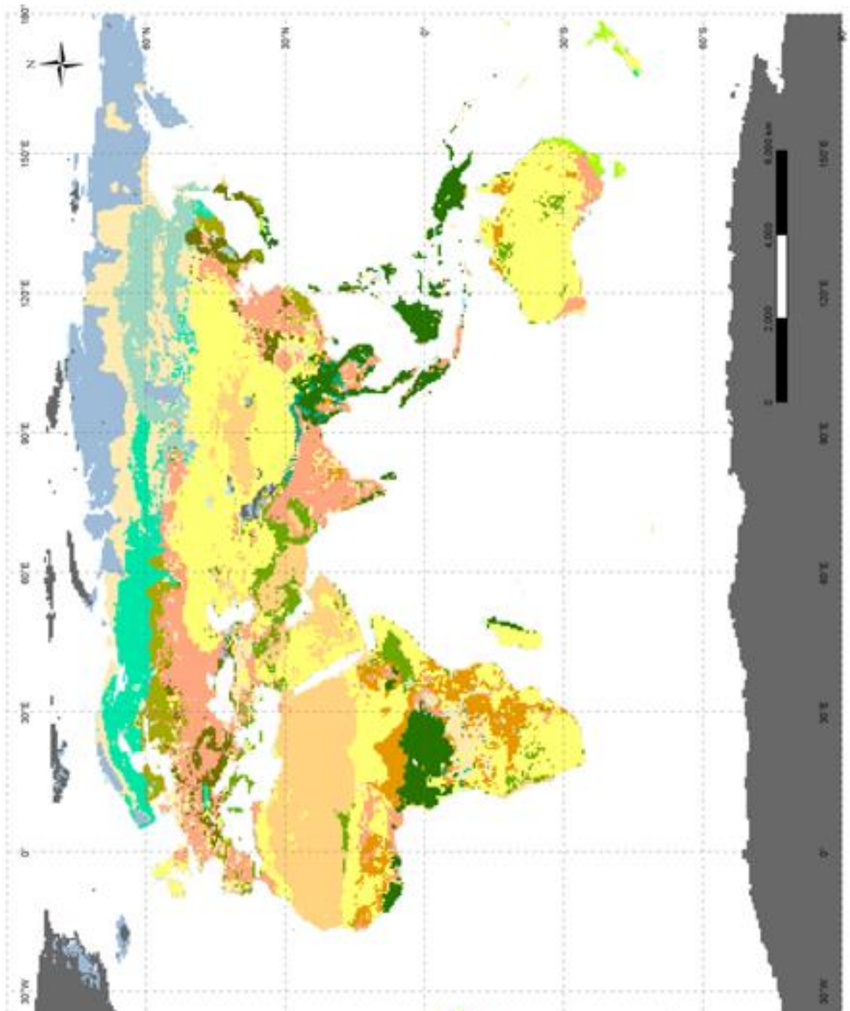


Figure 32: Global land cover in 2010 as estimated by the HYDE model

The effect of industrialization can be shown by rising income levels after 1825 in Europe and Japan, illustrated by the gross domestic product per capita for selected nations (Figure 33). Over the same period of economic development and rising incomes the area of forested land bottoms-out and then steadily increases (Figure 34). This would tend to support the forest transition hypothesis.

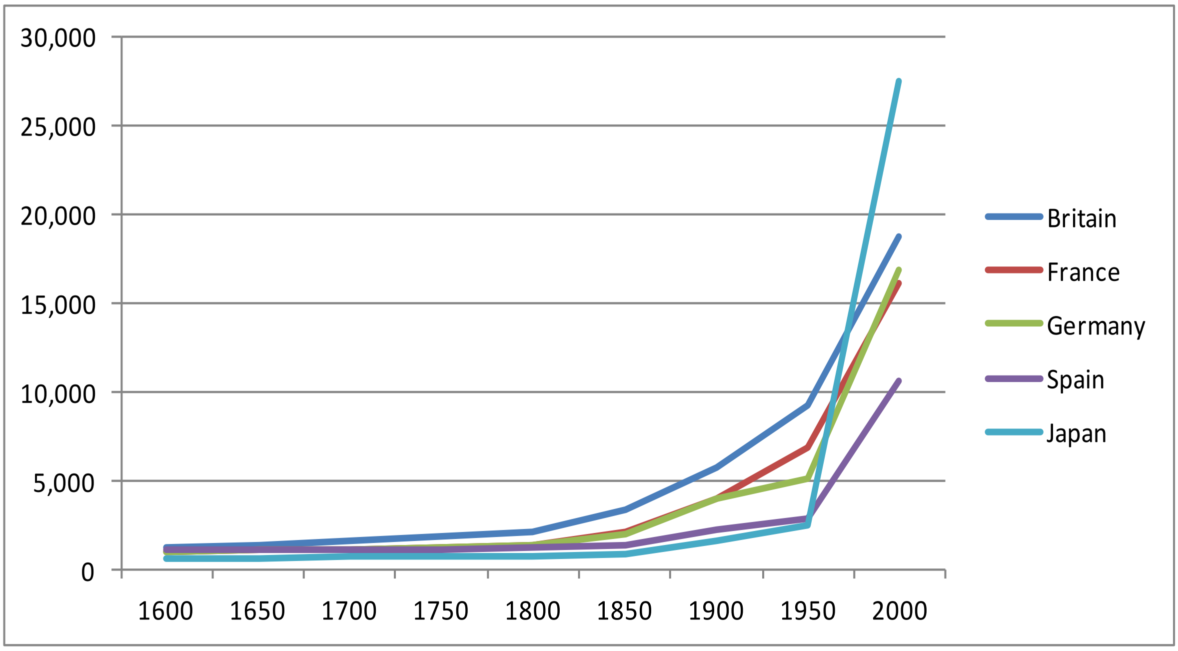


Figure 33: GDP at purchasing power parity per capita (US$ year 2000), 1600 - 2000

Source: Data from Contours of the World Economy, 1–2030 AD: Essays in Macro-Economic History, by Angus Maddison, Oxford University Press, 2007, ISBN 978-0-19-922721-1, p. 382, Table A.7.

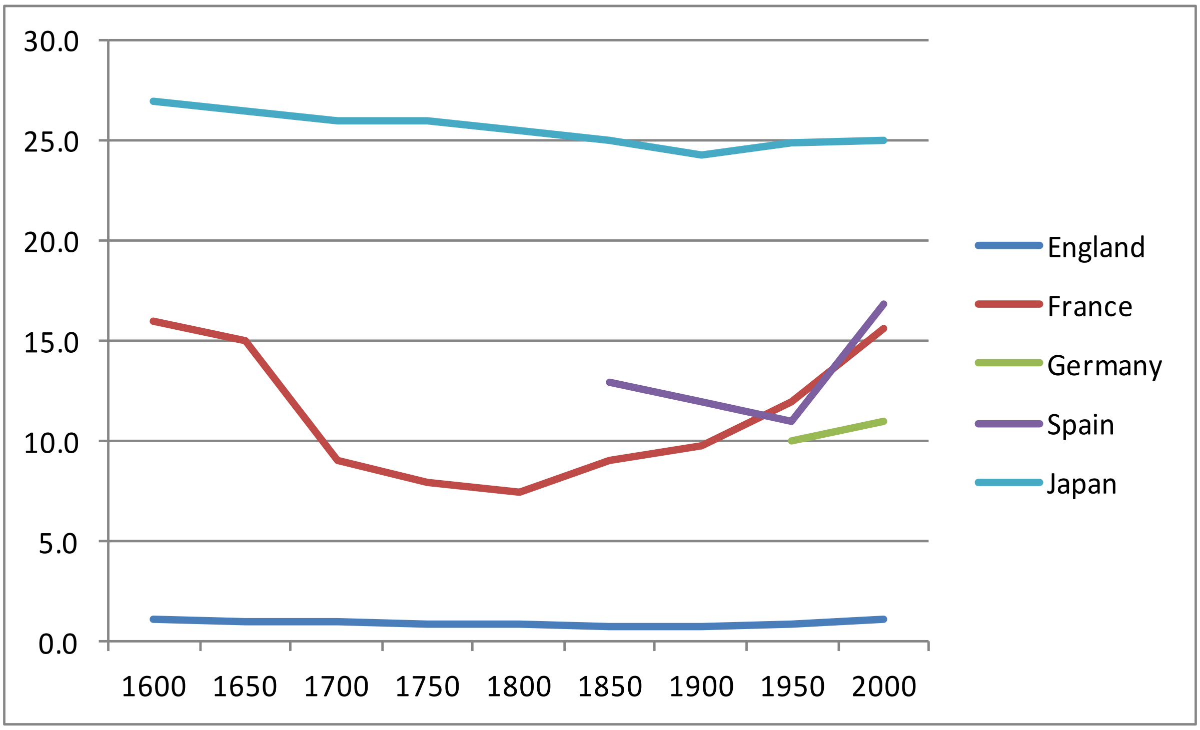


Figure 34: Forest area (millions of hectares), 1600 - 2000

Sources: FAO; Arte TV, 2012; SECF Sociedad Española de Ciencias Forestales, 2010; Driver, 2012.

However, whilst the forest transition curve in Europe is likely to be the outcome of various factors (e.g. the historical switch in energy sources following new discoveries thus reducing demand for firewood and charcoal), an important factor could be off-shoring deforestation. From early in the last century the location of forest loss shifted from temperate regions to the tropics, both to source timber and for additional crop land, initially driven by colonialism (for example, up to 33 million hectares of forest were cleared in India during the most intense colonial exploitation of wood resources (1850 - 1920) (Williams, 2002)). However, this has continued up to the present time as tropical nations have exported agricultural commodities as a means of earning foreign currency and raising revenues. FAO (2001) estimate that 14.6 million hectares of natural forest was lost each year over the 1990s, with 97% of this being tropical forests.

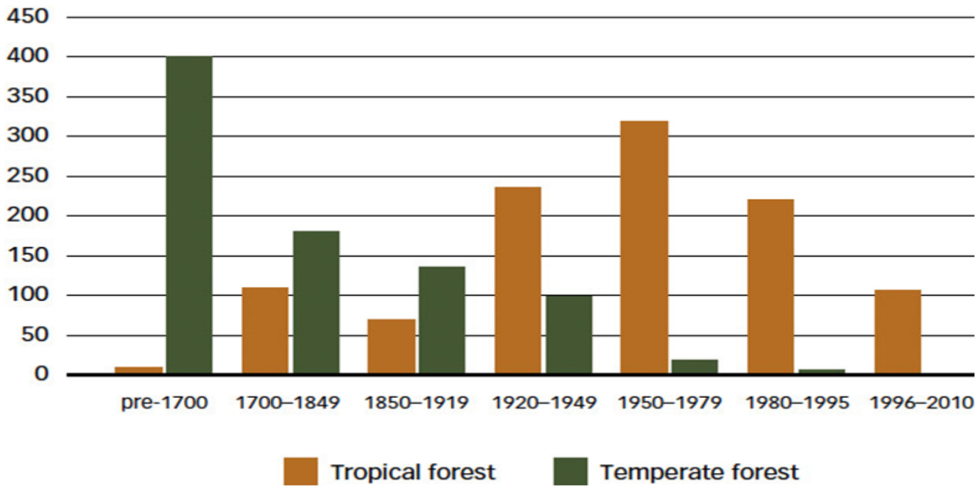


Figure 35: Deforestation by type of forest (million hectares)

Source: FAO (2012).

Deforestation for tobacco plantations in Sumatra (circa.1900). Tropenmuseum, Amsterdam.

The USA saw significant levels of deforestation following European settlement; but over the last century the forest area has increased (mainly in the east), so that it now has roughly two-thirds of the original (i.e. 1620) cover (FAO, 2012). This has occurred even while population has increased and the economy grown. It is likely to be partly a result of the intensification of agriculture allowing increased production from a smaller area of land most suited to large-scale agriculture (FAO, 2012). This seems to support the theory of a forest transition curve. However, a further factor is that net imports of wood and wood products have increased over the last several decades, so that almost one-third of demand is currently met by imports (USDA, 2014). It is worth noting that the quality of the forest re-growth (in terms of age structures and biodiversity) over the last century may be lower than the original forest. In a new development, some of the regenerated natural forests in the USA are being harvested (in addition to plantation forests) in order to produce pellets for biomass power stations. Some of these are exported - the USA eastern region has become the largest exporter of wood pellets in the world, with over 5 million tons of wood pellets forecast to be exported in 2015 (Hammel, 2013). This suggests that forest (at least in terms of area of mature forest) might begin a new downward trajectory in the USA if the harvesting rate further increases significantly.

Figure 36: Current US forest cover (left), compared with forest cover at its lowest point (right)

Source: current forest cover map by Robert Simmon (NASA, no date); historic map from Meyer (1995).

Around two-thirds of China is estimated to have been covered in forests 4000 years ago, but by 1949 forest cover had fallen below 10% (FAO, 2012). Forest cover currently stands at 22%, but has been achieved partly by increasing food and wood imports from other nations (Nengwen, 2012), which may be off-shoring land conversion (EIA, 2012). The situation is similar in Japan. Whilst Japan also experienced periods of rapid population and conversion of forest to agricultural land, the benefits of forest management and conservation were recognised earlier (because most of the forests were in mountainous areas), and planting over the last two centuries expanded the forest area by millions of hectares to nearly 70% of Japan’s total land area (FAO, 2012). However, this situation was aided by the ability to import wood products (and increasingly food), partly displacing the deforestation footprint on to others (Global Witness, 2014).

In South America a number of the largest intact natural forests still exist, and these were largely untouched as recently as the middle of last century[40]. For instance, using satellite (Landsat) images from the early 1970s, the forest cover of Paraguay's Atlantic Forest Ecoregion was mapped (Huanga et al., 2007). In 1973 almost three-quarters of the region was covered by forest, but this was quickly reduced to 40% by 1989 and then to 25% by 2000 (Huanga et al., 2007). A significant part of the

[40] Latin America was probably three-quarters forested before European settlement, and today it is around 50% (FAO, 2012), with much of the deforestation occurring in the twentieth century (Williams, 2002).

Brazilian Amazon (in the eastern and south-eastern part of the Amazon known as the Arc of Deforestation) was also deforested over the period 1970-2000 (Fearnside, 2005).

However, halting or reversing deforestation over recent years has not been confined to the richer industrial nations (who have, to some extent, displaced their footprints). During the period 2005 – 2010, about 80 countries (including some of the poorest) reported either an increase or no decline in forest area. Countries reporting increased forest area include Costa Rica, Cuba, India, Morocco, the Philippines, Rwanda, Tunisia and Uruguay (FAO, 2012).

Costa Rica is perhaps the most dramatic example of the reversal in land-use-change patterns. But rather than a validation of the forest transition hypothesis, Costa Rica is perhaps an example of a reorienting of the economy, and demonstrates how national economic growth does not have to be driven by deforestation if alternative options are explored. The national forests, often adjoining beaches, have made the country a popular destination for eco-tourists, with tourism bringing in almost US$2 billion a year[41] (Travel Smart, 2014). Although Costa Rica was once 99% forested, forest cover steadily diminished to around a third of the land area. However, after 1997 forest cover stabilized and then increased as reforestation occurred in a number of areas (Figure 37). This occurred at a time (1997 - 2005) when the population increased by over 18% and the economy by 47.5% (World Bank, 2014a).

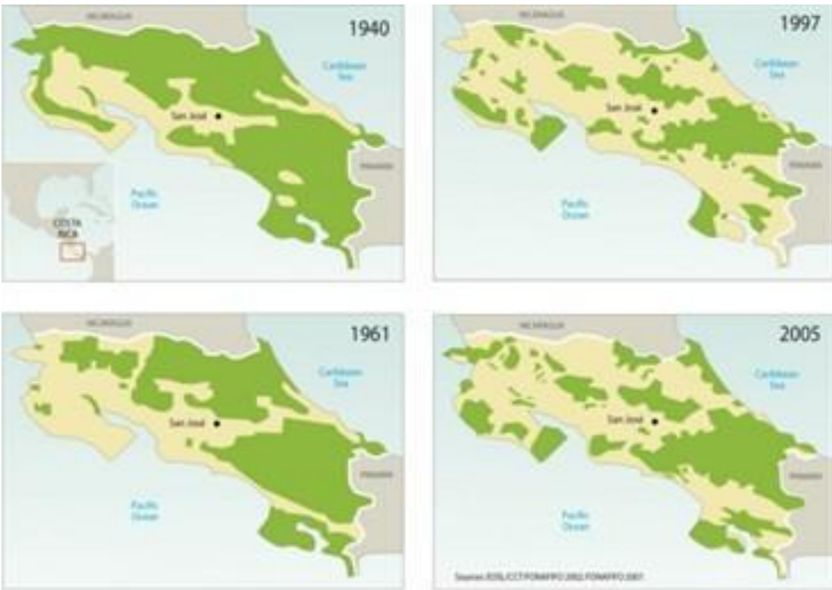


Figure 37: Costa Rica's historical forest cover (1940 – 2005)

Source: UNEP 2009. Note: The original cartography was undertaken by Costa Rica’s National Forest Financing Fund, FONAFIFO.

A key question that arises is: what happens when there are no new untouched territories to expand into (i.e. to displace deforestation on to)? Does the forest transition theory work at a global scale? Even if it did, another issue to be considered is the distribution of the remaining forest and what this means for the delivery of ecosystem services. Some estimates of original global forest cover exist,

[41] i.e. equivalent to US$400 / year for every Costa Rican.

which give an indication of overall loss of forests[42], but the pattern is complex, as in some areas forest re-growth and plantations have increased forest cover. Europe still had 80% forest cover around 2000 years ago, but now forests occupy only one-third of the land mass (FAO, 2012). Not only is there no indication of whether this level is adequate for the provision of key ecosystem services, but the distribution of the forest across the landscape (i.e. in fragments or just in remote areas) may not be optimal for ecosystem service provision, with some areas having close to no tree cover. Since much of the increase in tree cover may be due to plantation forest (sometimes with a single non-native species)[43], the forest cover statistic can be misleading in ecological terms. Therefore, a forest transition may mask the fact that the lost ecosystem services associated with the original deforestation have not recovered to the same extent as forest land cover[44].

The above raises concerns about the likelihood of a global forest transition occurring automatically as a result of economic growth in developing countries, let alone whether this would maintain biodiversity and ecosystem service provision. However, there are a number of countries that have not reached high levels of income but that have managed to maintain or increase forest cover. Therefore, the forest transition theory may be unhelpful when exploring economic development options, and instead the adoption of a more proactive Green Economy strategy would be preferable if sustainable development is the policy aim.

Whilst the underlying driver of global deforestation can be traced back to increased demand for raw materials[45] and associated global policies, the direct drivers can vary, and operate in the context of national policy incentives for land use, the strength or otherwise of conservation policies and law enforcement, and the suitability of land for conversion to agriculture. A recent meta-analysis of deforestation drivers examined the twenty most commonly studied variables associated with deforestation, using a database of 117 spatially explicit econometric studies of deforestation published in peer-reviewed academic journals from 1996 – 2013 (Gallon and Busch, 2014). It found that forests are more likely to be cleared where economic returns to agriculture are higher, either due to more favourable climatic and topographic conditions, or due to lower costs of transporting products to market[46] (Gallon and Busch, 2014). Together with high commodity prices these factors combine to bring about economic incentives that make forest conversion appear more profitable than forest conservation.

[42] Goldewijk (2001) estimates global forest cover fell by almost 30% from undisturbed state up to 1990.
[43] Since the forest transition is based on land cover and not the quality of the forest, it also ignores forest degradation.
[44] Perhaps apart from the provision of timber.
[45] A function of both growing global population and growing average per capita consumption.
[46] Obviously the prices of agricultural commodities will have a large impact on the economic returns of particular crops.

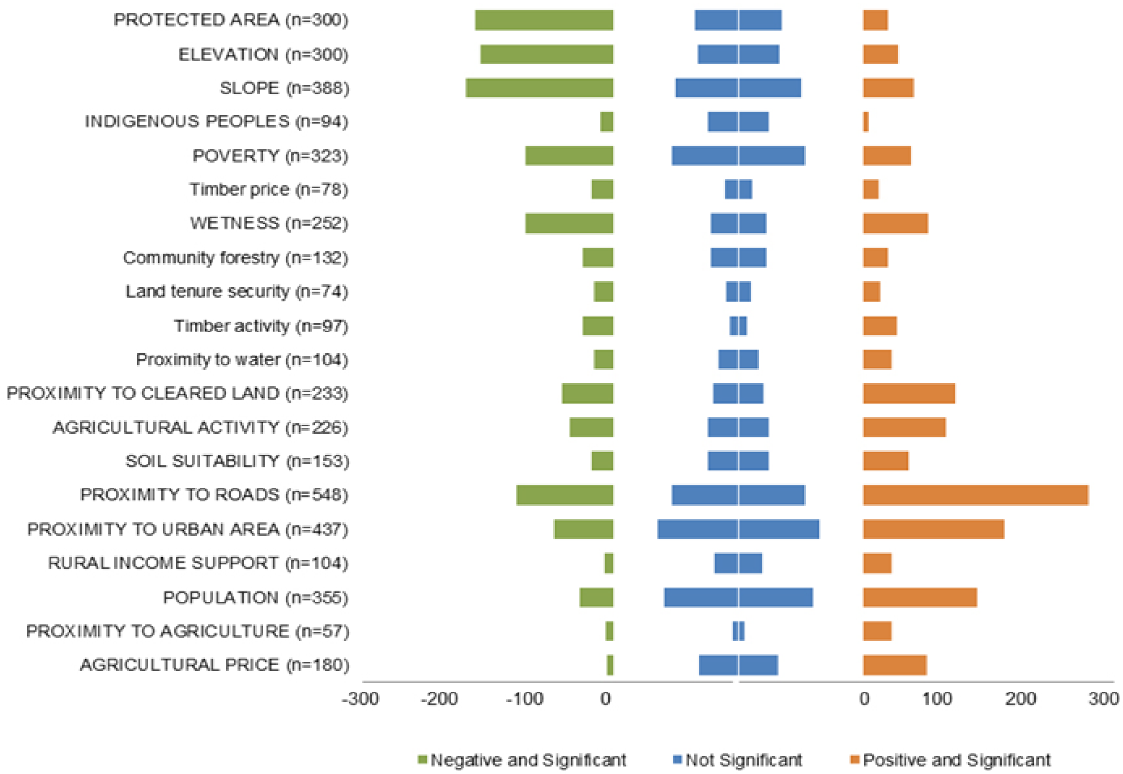


Figure 38: Deforestation drivers

Source: Gallon and Busch, 2014

Although individual countries may be able to both increase forest area and meet other needs such as growing their economy (as demonstrated by Costa Rica), internationally coordinated actions are likely to be required in order to achieve this at the global scale. A global forest transition curve appears unlikely without this coordinated action, since not only is there on-going demand for wood products but also a need to increase food production as the global population increases. This suggests that multiple sectors of the economy (as well as institutions) will need to be engaged in order to end net deforestation at the global scale. Demand reduction could be achieved through increased efficiency of wood use[47] (e.g. improved stoves in developing nations) and recycling of wood and wood products (such as paper and cardboard), as well as improved crop yields (and changes in the consumption of some foods) and more sustainably managed wood production (including forest plantations). This can be achieved by adopting a suite of policies, including offering financial incentives for demand reduction, and increasing the value of standing forests (i.e. reversing the economic incentive to convert forests). If the different deforestation drivers are not addressed simultaneously then action to reduce net deforestation is less likely to succeed. Bringing about a significant reduction in global forest loss is a challenging task that should not be underestimated.

[47] As well as composite materials.

8. Is there a link between deforestation and economic success?

Whilst traditional ideas of economic development might have centred on rapid conversion of natural resources to exportable raw materials and maximising the area of land under agriculture, this approach has been shown to have serious negative impacts that may result in a lower performing economy in the long term. An alternative, the Green Economy approach, has been proposed in response. UNEP defines a Green Economy as one that 'results in improved human well-being and social equity, while significantly reducing environmental risks and ecological scarcities (UNEP, 2014). Therefore, a Green Economy is low carbon, resource efficient and socially inclusive, employing the principles of sustainable production and consumption (UNEP, 2014). A Green Economy is more than just a low-carbon development pathway; it also considers natural resources more broadly, as sources of prosperity. Every country will have its own set of human and natural resources, and its own approach to a Green Economy (UNEP, 2014).

The governments of some developing countries may feel that they need to extract natural resources at a high rate to grow their economies. But such development is often unsustainable, i.e. it cannot be sustained in the long term, and the costs associated with it in terms of impacts on the functioning of ecosystems providing services, mean that at some point this will become uneconomic (the costs to the country exceeding the benefits, and actually becoming a drag on economic development). Obviously some use of natural resources is required and the Green Economy approach recognizes this. Strategic investment of the gains made from natural resource use should be invested in renewable energy, material efficiency improvements, as well as conservation and restoration of natural infrastructure (ecosystems delivering key services) to off-set the depletion of natural resources. The transition to a Green Economy will require cross-sectoral planning and resource management, and market signals that give appropriate values to ecosystem services (UNEP, 2014). Green Economy activities in relation to forest retention would include promoting forest protection whilst improving livelihoods, such as sustainable NWFP harvesting accompanied by processing and adding value, developing forest plantations as an alternative source of wood products, and encouraging nature-based tourism in forests.

Although no countries presently claim to have completely transitioned to a Green Economy, a number are in the process, and there are examples of countries protecting their natural capital (especially forests), whilst growing their economies. The following five countries have over 50% forest cover[48], and are also in the top 40% of countries in terms of income per capita (two of them are in the global top 10% of income per capita): Estonia, Finland, Republic of Korea, Slovenia and Sweden. They represent a mixture of high, low and medium population densities. They provide practical examples of modern growing economies which have managed their development at the same time as having a high forest cover, in the case of the Republic of Korea restoring its forests from near-devastation forty years ago to today's 64% forest cover. Whilst the Republic of Korea has managed this by being able to import large quantities of food, this is not the case for all of these countries. An analysis of Finland found that although imports of agricultural products and raw materials for food industries are increasing, self-sufficiency (as measured by the difference between imports and exports) was

[48] It is worth noting that Argentina now has only 10-14% forest cover but is continuing to deforest.

estimated at no larger than 233,000 ha, in comparison to 2.3 million ha of Finland currently in active farming (Sandströma et al., 2014).

Table 38: High income per capita countries with high forest cover

Country	GNI per capita (US$ 2013)	% tree cover (in year 2000)
Sweden	59,240	61
Finland	47,110	64
South Korea	25,920	53
Slovenia	22,750	64
Estonia	17,370	56

A number of less developed countries have also managed to halt deforestation. For example, 15 years ago the villages around Abrha Weatsbha in northern Ethiopia were on the point of being abandoned as the hillsides were barren, with the soil being eroded and food production declining due to floods and droughts (Vidal, 2014). However, following the planting of millions of tree seedlings and closing off large areas to grazing (allowing natural regeneration), soil quality and hydrological functioning has improved (Mekuria, 2013). Across Ethiopia several hundred thousand hectares are now under 'exclosures', and the country has pledged to undertake forest landscape restoration over a further 15 million hectares. Other recent commitments for forest landscape restoration have come from Uganda (2.5 million hectares), Democratic Republic of the Congo (8 million hectares), Colombia (1 million hectares), Guatemala (1.2 million hectares), and Chile (100,000 hectares). Already in Burkina Faso over 200,000 hectares of land has been re-greened, whilst at the same time food production has grown enough to feed an extra 500,000 people (Vidal, 2014).

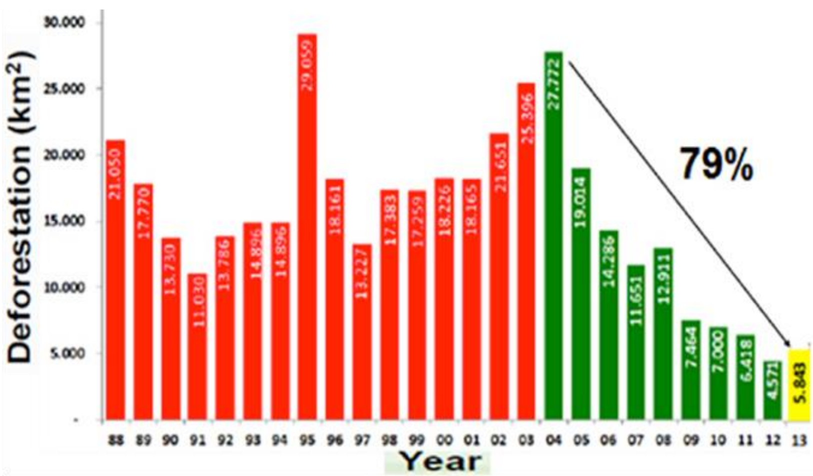


Figure 39: Deforestation rates in the Brazilian Amazon 1988-2013

Source: UNFCCC, 2014 (data from INPE).

A sharp decline in deforestation in the Brazilian Amazon between 2004 and 2012 (Figure 39), whilst at the same time increasing agricultural output, suggests that it is possible to halt the advance of a vast agricultural frontier. The reasons for the reduction in deforestation appear to be enforcement of laws, interventions in soy and beef supply chains, restrictions on access to credit, and expansion of protected areas (Nepstad et al., 2014), as well as accurate, transparent deforestation monitoring and high-level political support. Analysis of approaches more widely suggest that promising means for stopping deforestation include reducing the intrusion of road networks into remote forested areas; targeting protected areas to regions where forests face higher threats; and tying rural income support to the maintenance of forest resources through payments for ecosystem services (Gallon and Busch, 2014).

REDD+ programmes may offer additional assistance for developing countries wishing to protect their at-risk forests whilst also aiding economic development. REDD+ offers the potential for payments for reductions in carbon emissions and enhancement of removals. If successful, REDD+ could help change the pattern of investment, and incorporate natural capital, poverty alleviation and ecosystem services into economic decisions.

An integral part of REDD+ is the implementation of good governance (through safeguards). Some countries have low levels of corruption and good security of land tenure, which will be associated with a stable agricultural sector, as well as the ability to police illegal logging and timber trade. This can help bring about low deforestation rates. Other countries are less fortunate, and here attempting to implement a forest conservation approach is unlikely to deliver positive results on its own. A prerequisite for a successful REDD+ is a strong and legitimate government with a sustainable land-use policy, and an independent police force and judiciary where the rule of law is widespread[49], and aiding this will be low levels of corruption throughout society. The need for good governance in building a sustainable economy extends beyond REDD+. Previous studies have concluded that countries that are less democratic tend to formally protect less land (Vincent et al., 2014). Analysis of the benefits from deforestation can identify the economic drivers, but this does not explain why deforestation is occurring, which is ultimately a function of the social drivers. Where there is widespread corruption and weak law enforcement the economic gains can be appropriated by a small minority at the expense of the population as a whole. Therefore, there will be a strong individual incentive to appropriate land, extract the resources and convert the land to the most financially profitable short-term use. Given that countries with less corruption and where the rule of law is instituted are likely to have a more successful economy; controlling deforestation is likely to be an indicator of future economic prosperity.

[49] This includes neutral armed forces, who also follow the rule of law.

9. Conclusions

This study has attempted to explore the economics of deforestation in order to determine economic drivers and to ascertain whether economic success necessitates deforestation. As part of this analysis, a methodology was developed to estimate the value of the ecosystem service costs of deforestation, frequently ignored by economic planners.

The case study of northern Argentina found that the costs of deforestation have been substantial - somewhere between a quarter and the total of the gains realized from deforestation between 1960 and 2010. The marginal gains in the most recent decade are lower as a result of rising environmental costs. It was not possible to value all of the costs, but the biggest costs that were valued relate to increased flooding, droughts[50] and future climate change impacts. These costs will be felt across the population of Argentina, whilst the direct benefits from the conversion of forests to agricultural land have accrued to a minority. However, agricultural export taxes do represent a significant portion of the country's revenue. This development path may be unsustainable in the longer-term, especially if soil degradation and droughts persist (thus lowering crop yields). An alternative development strategy would be to limit further forest loss. A detailed spatial panning approach, mapping costs and benefits of land conversion, would assist in enhancing economic gains from land management.

Additionally, this study has posited that the forest transition theory is possibly outdated[51] and not always useful when considering sustainable development. Firstly, the hypothesized change in forest cover as economies develop has been realized by partly shifting the deforestation footprint onto other countries. Secondly, forest growth in some countries occurred at the same time as a number of global developments in technology that reduced the per capita demand for both land and wood (and wood products). Further, the transition theory says little about the quality of the increased forest cover in terms of delivering ecosystem services. An additional factor is that some developing countries are now realizing that it makes economic sense to protect and restore forests[52].

Forest protection may occur in conjunction with economic growth under the Green Economy approach, and REDD+ offers a means for developing countries to access additional finance in order to help conserve forests. However, much more work is required to substantially reduce global deforestation. One key action is to further promote the understanding of the costs of deforestation to policy-makers and the wider public. As part of this endeavour, it is hoped that REDD+ countries will consider adopting and improving on the approach trialled in this study to evaluate the economic costs of deforestation.

[50] Further work is required to determine the exact role of different forest types across Argentina in hydrological functioning.
[51] At the very least, we suggest that further, more detailed, analysis of the forest transition theory is required.
[52] Increasingly with the opportunity of support from donors.

Appendix A: CICES classifications

CICES for ecosystem accounting			
Section	**Division**	**Group**	**Class**
Provisioning	Nutrition	Biomass	Cultivated crops
			Reared animals and their outputs
			Wild plants, algae and their outputs
			Wild animals and their outputs
			Plants and algae from in-situ aquaculture
			Animals from in-situ aquaculture
		Water	Surface water for drinking
			Ground water for drinking
	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing
			Materials from plants, algae and animals for agricultural use
			Genetic materials from all biota
		Water	Surface water for non-drinking purposes
			Ground water for non-drinking purposes
	Energy	Biomass-based energy sources	Plant-based resources
			Animal-based resources
		Mechanical energy	Animal-based energy
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals
			Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems
			Dilution by atmosphere, freshwater and marine ecosystems
			Mediation of smell/noise/visual impacts
	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates
			Buffering and attenuation of mass flows
		Liquid flows	Hydrological cycle and water flow maintenance
			Flood protection
		Gaseous / air flows	Storm protection
			Ventilation and transpiration

	Maintenance of physical, chemical, biological conditions	Lifecycle, habitat and gene pool protection	Pollination and seed dispersal
			Maintaining nursery populations and habitats
		Pest and disease control	Pest control
			Disease control
		Soil formation and composition	Weathering processes
			Decomposition and fixing processes
		Water conditions	Chemical condition of freshwaters
			Chemical condition of salt waters
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations
			Micro and regional climate regulation
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in different environmental settings
			Physical use of land-/seascapes in different environmental settings
		Intellectual and representative interactions	Scientific
			Educational
			Heritage, cultural
			Entertainment
			Aesthetic
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Spiritual and/or emblematic	Symbolic
			Sacred and/or religious
		Other cultural outputs	Existence
			Bequest

Appendix B: Estimation of the historic cost of carbon

The following lays out the theoretical underpinnings for the calculations of the social cost of carbon (SCC) used in this study. Global CO_2 emissions accelerated in the first half of the nineteenth century as exploitation of fossil fuel reserves became more common, fuelling industrialization, first in Britain and then the rest of Europe, followed by the US and Japan, and then the other nations of the world. In the first decade of the twentieth century, emissions from fossil fuels overtook land-use change emissions for the first time (Figure 40).

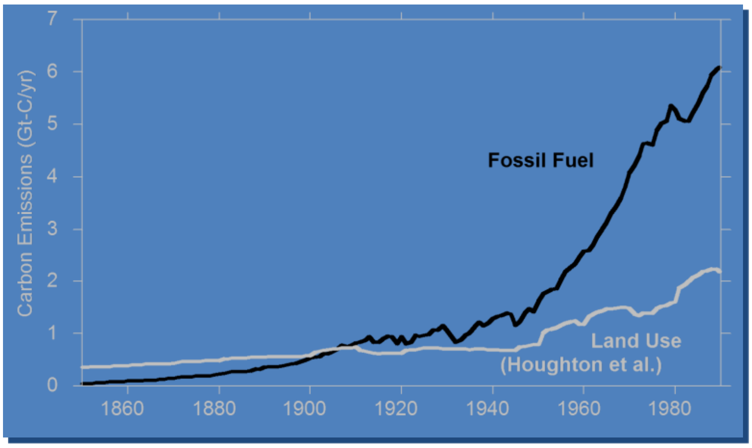


Figure 40: Global carbon emissions, 1850-1990

Source: Ramankutty (2001).

These combined fossil fuel and land-use emissions have resulted in an increase in atmospheric concentrations of CO_2 since the year 1800 above levels that existed at any point over the previous several centuries (Figure 41).

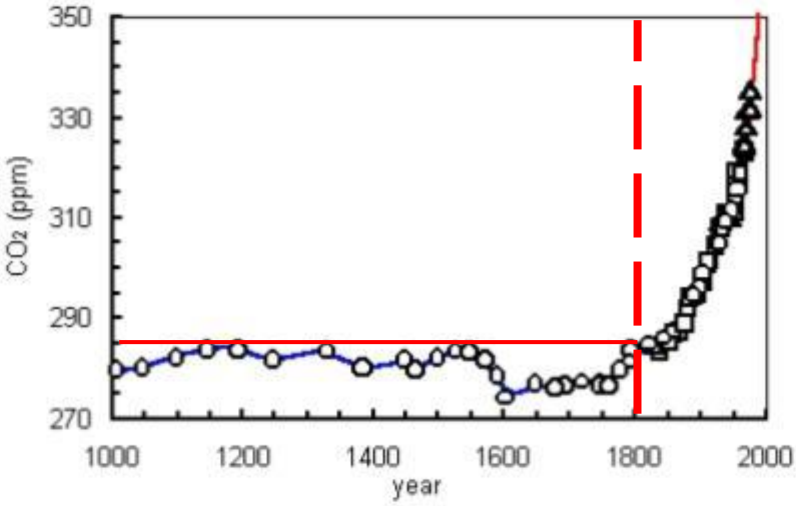


Figure 41: Ice core data showing CO_2 levels, year 1000-2000

Source: CSIRO, http://www.dar.csiro.au/pub/info/greenhouse.html (red lines added)

As a result of the increased levels of CO_2 in the atmosphere, the Earth's natural greenhouse effect has been amplified so that more energy is trapped in the atmosphere, leading to global warming which has resulted in the early stages of climate change. Already, historic emissions have increased temperatures above the 20th century mean. For example, June 2014 was the 352nd consecutive month with a global temperature above the 20th century average (NOAA, 2014). In addition to higher temperatures leading to drought and extreme heat events, increased energy in the atmosphere also results in increased storm intensities and extreme precipitation events.

Climate change is already disrupting the flow of the jet stream (the jet stream consists of ribbons of strong winds reaching speeds of up to 320 km per hour, located 9-16 km above the surface of the Earth, which move weather systems around the globe), amplifying circulation waves high in the atmosphere. As a result summer heatwaves and downpours have become more frequent in the northern hemisphere this century (e.g. droughts in the US, the heatwaves in Europe in 2003 and Russia in 2010, and rains that caused flooding of the Elbe and Danube Rivers in Europe in 2002, and repeated flooding in the UK over the last decade) (Coumoua et al., 2014). In addition to damaging crops, houses and infrastructure, some of these impacts are on natural assets, such as forests. For example, in August 2014, in the wake of the highest temperatures Sweden has experienced on record, the country experienced the largest forest fire of the last four decades, destroying an area of forest 10 km by 15 km (Stallard, 2014) and leading to government emergency aid of US$44 million (Molin, 2014). The number of floods, storms and extreme temperature events recorded have multiplied five-fold over each of the last four decades (Figure 42) (though some of this may be due to urban growth and better reporting). According to Swiss Re (2014) there are already costs being felt as a result of climate change impacts, and these costs are increasing[53].

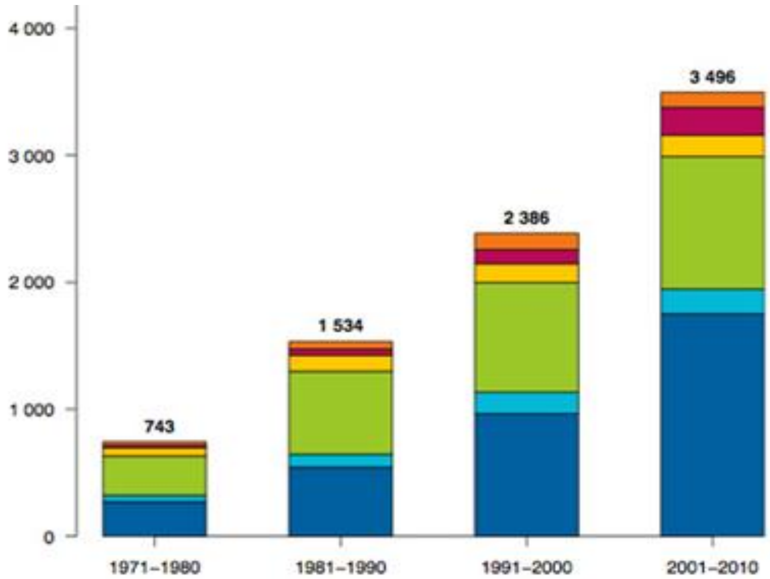


Figure 42: Number of disasters across the globe each decade by hazard type

Dark blue = floods. Light blue = mass movement wet (subsidence, rock falls, landslides). Green = storms.
Yellow = drought. Magenta = extreme temperature. Orange = Wildfires

Source: World Meteorological Organization (2014)

[53] Though again some of the increased costs will be due to growing wealth and urban expansion, reduced retention due to the straightening of water courses, the construction of weirs, the loss of water meadows and wetlands, and increased surface sealing.

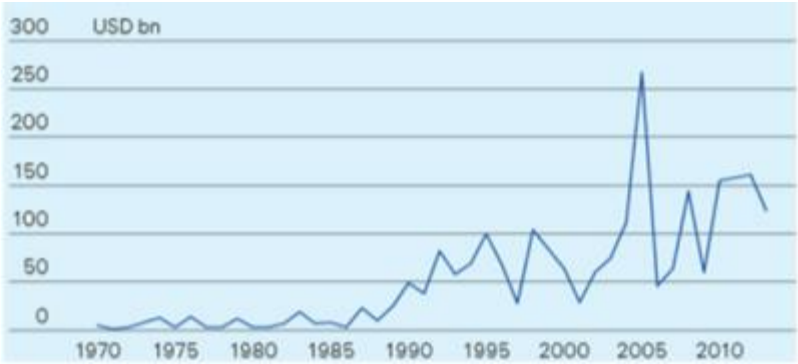


Figure 43: Economic losses from extreme weather events, 1970–2013

Source: Swiss Re (2014)

Each extreme weather event is associated not only with direct economic losses, but also impacts on economic development. Whilst some have hypothesized that natural disasters can stimulate economic growth, or that short-run losses quickly disappear, recent research has found that national incomes decline relative to their pre-disaster trend and do not recover within twenty years (Hsiang and Jina, 2014). Both rich and poor countries exhibit this response, with a major event (90th percentile cyclone) reducing per capita incomes by over 7% two decades later[54], which is effectively undoing over three and a half years of average development (Hsiang and Jina, 2014).

The first step in assessing the impacts of forest loss on the economy via climate change is to establish the mechanisms by which forest loss has an effect on climate. The most obvious pathway is through the CO_2 impacts of deforestation. However, forests influence the climate through a variety of mechanisms including albedo[55], emissions of biogenic volatile organic compounds[56] and cloud formation[57]. As there is currently much uncertainty over the total net impacts of these various factors, they are not considered further here. That is, it is assumed that the positive and negative warming impacts, excluding CO_2, largely cancel out one another in tropical regions. This assumption should be subject to review as our understanding in this area develops. Thus the sole climate factor considered in this study is the net change in CO_2 resulting from deforestation, and so a value for CO_2 must be estimated.

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[54] This is on a par with a banking crisis, and over twice the impact of a civil war (Hsiang & Jina, 2014).

[55] There can be a net cooling impact as a result of land use change from forest to cereal crops and pastures due to changed albedo (i.e. the fraction of solar energy reflected from the Earth back into space). However, some crops (such as rubber plantations) may have similar albedo to natural forests.

[56] Vegetation emits chemicals (BVOCs - biogenic volatile organic compounds), and some of these compounds can produce a warming effect and others a cooling effect. The levels of BVOCs produced depends on the tree species, temperature and light levels among other variables. Since many replacement crops in tropical regions (e.g. oil palm) also give off these compounds, there may be no net change in these effects. Additionally, agricultural land management can produce other GHG emissions (e.g. nitrous oxides resulting from tillage and fertilizer use on arable land, or methane emissions from rice paddies and cattle on pastures).

[57] Low clouds have a cooling effect (by reflecting solar energy) and tropical forests can have high evapotranspiration rates, promoting low-level cloud cover.

The Social Cost of Carbon (SCC) is the present discounted[58] value of the future stream of costs resulting from today's emission of a new unit of CO_2. Current CO_2 emissions will have a negative impact in the future, for which costs can be estimated (social costs are also called damage costs). The present discounted value results in the situation where the further into the future a cost occurs, the less value it is given in present terms (see section 6.6 for more on discounting).

All CO_2 emissions since the year 1800 have added to the atmospheric CO_2 concentration. Historic (pre-2010 in this instance) CO_2 emissions are already having an impact on climate and causing economic losses. Therefore, these past emissions also have an associated cost. Emissions before 1800 could be assumed to have no cost as the sink was broadly in balance and CO_2 levels in the atmosphere were stable; though there is some evidence that CO_2 emissions had increased over several millennia driven by human actions (Boyle et al., 2011), making the planet warmer than it would have been without anthropogenic influence (Kaplan et al., 2011). Such warming (up to 1800) could be described as having a net benefit rather than cost if the climate had been on a cooling trajectory (thus the SCC function could be U-shaped).

There is no literature available on historic costs of CO_2 emissions, apart from estimates of the SCC over coming decades which were carried out in the 1990s, and so cover the intervening years (however, as our understanding of the impacts of climate change have improved these estimates have become out-dated). Ideally the cost of the CO_2 emissions from 1800-2010 would be calculated based on the past, current and future damages that are associated with them. However, this would be a substantial task and a different approach will have to be adopted here.

Most of the aggregate impacts reported by the IPCC (1996) consisted of monetized valuations of the likely damage that would be caused by a doubling of CO_2 concentrations. For developed countries, estimated damages were of the order of 1% of GDP. Including impacts on the environment and human health, and adjusting for evidence of greater climate sensitivity, increased this to 14% of GDP (IPCC, 2007b). Early calculations of the SCC ranged from US$5 - $125 per tonne of carbon (in 1990 dollars) (Parry et al., 2007). Subsequent estimates did little to reduce this range (Table 39).

Table 39: Estimates of the Social Cost of Carbon (SCC)

Authors:	**Year:**	**SCC estimate (US$ per tonne of carbon):**	**Notes:**
Clarkson & Deyes	2002	central value of $105 (in year 2000 prices)	range of $50 - $210
Pearce	2003	$4 - 9	
Tol	2005	median of $14, a mean of $93	95th percentile estimate of $350
Stern	2007	$310	

[58] A reason for 'discounting' future costs and benefits is human's time preference, which refers to the desire to enjoy benefits in the present, while deferring any negative effects of doing so. A discount rate is applied to costs and benefits occurring in the future, and so, because of compounding, values in the far distant future become very small unless the discount rate is a very low one (the UK Treasury notes that for periods greater than 300 years a 1% discount rate should be used).

The range reflects uncertainties in climate and impacts, coverage of sectors, and choices of key variables, such as:

i. the damage sensitivity to temperature,
ii. the discount rate, and
iii. equity weights (used to aggregate monetized impacts over countries).

Stern (2007) calculated a mean estimate of the SCC in 2006 of US$310 per tonne of carbon, largely due to the adoption of a low rate of time preference. Whilst the IPCC adopts a discount rate of 2.4% there have been on-going discussions in the climate change economics literature as to what discount rate should be adopted (some have even advocated a negative rate). Pure-time discounting is essentially discrimination by date of birth (Dietz and Stern, 2014). This is in the sense that a life, which is identical in all respects (including time patterns of consumption) but happens to start later, is accounted as having less value. For instance, a person born in 2050 has a life only “worth” half as much as someone born in 2015, using a 2% discount rate. It should be noted that when analyzing climate policies the rate of future economic growth is considered alongside time preferences in order to determine the discount rate.

Estimates of the social costs of carbon reflect an incomplete subset of relevant impacts, as many significant impacts have not yet been evaluated; so overall, it is likely that current SCC estimates are understated. However, a number of governments have published estimates of the SCC for use in economic planning and environmental rule making. The US EPA has calculated the SCC over the coming decades comparing the results of multiple discount rates (two are shown in Table 40).

Table 40: US EPA estimates of the SCC for both carbon and carbon dioxide

	SCC at discount rate (2011 US$/t C)		**SCC at discount rate (2011 US$/t CO2)**	
Year	**5% Average**	**2.5% Average**	**5% Average**	**2.5% Average**
2015	44	224	12	61
2020	48	250	13	68
2025	55	272	15	74
2030	62	294	17	80
2035	73	312	20	85
2040	81	338	22	92
2045	95	360	26	98
2050	103	382	28	104

Source: Carbon values calculated from values for CO_2 from the United States Environmental Protection Agency; The Social Cost of Carbon http://www.epa.gov/climatechange/EPAactivities/economics/scc.html

A simplified approach to estimate historic SCC values is adopted here. It is assumed that the damages already experienced and that will occur in future, due to CO_2 emissions up to the year 2010, will have a damage value (and resulting carbon price) which follows the trajectory of future prices up to 2050. It is assumed that CO_2 emissions before the year 1800 have no impact and thus zero cost, and that above the year 1800 level they have a net negative impact (though an almost infinitesimally small one for the first tonne of CO_2 above this level). Thus the existing US EPA SCC figures for 2015 are projected backwards in time. Clearly, the approach for estimating carbon costs could be improved, and further research could be conducted in this area.

The results of the back-projection are as follows. For the 5% discount rate, in 1965 the SCC falls below US$1 t/$CO_2$, and by 1800 it becomes less than a cent (US$0.0002 t/$CO_2$). As the year 1800 is posited as the cut-off year before which CO_2 emissions were in balance with carbon sinks, the pre 1800 emissions are assumed to be valued at zero (Figure 44).

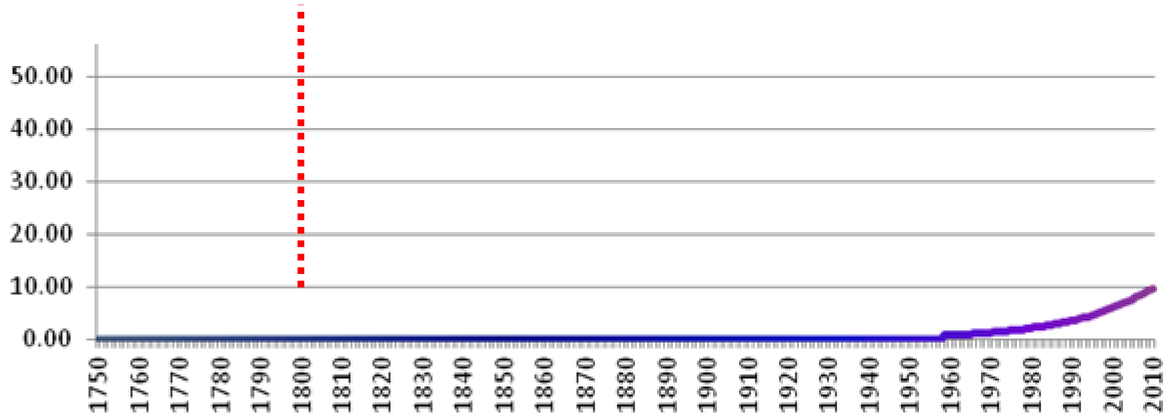

Figure 44: Social Cost of Carbon backward projection using US EPA figures based on 5% discount rate (2014 US$/t CO2)

With a lower discount rate the SCC doesn't fall below US$1 t/$CO_2$ until 1850, and by 1800 it is still a little above zero, at US$0.28 t/$CO_2$. The 1960 to 2010 costs per tonne are also significantly higher.

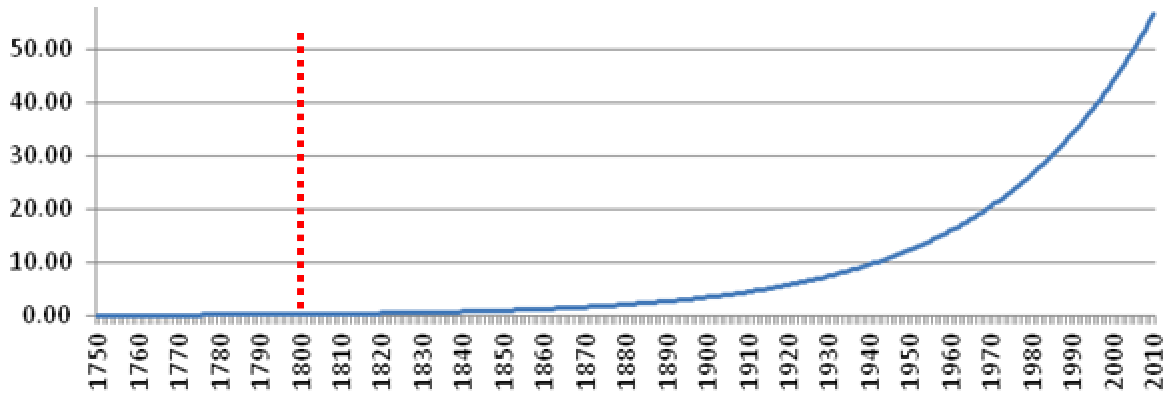

Figure 45: Social Cost of Carbon backward projection using US EPA figures based on 2.5% discount rate (2014 US$ t/CO2)

The rounded average (mean) cost of CO_2 per tonne averaged over each decade is as follows:

Table 41: Rounded estimates for the SCC in past decades, by back-projection of US EPA values

US EPA 2.5% (2014 prices) Average decade SCC	US$ per metric t of CO2
1960s	18
1970s	23
1980s	30
1990s	39
2000s	50

US EPA 5% (2014 prices) Average decade SCC	US$ per metric t of CO2
1960s	1
1970s	2
1980s	3
1990s	4
2000s	7

An alternative approach to estimating the SCC was also taken. Using a point estimate for climate damages in 2010 (DARA, 2012), these were projected back using an exponential function[59], and post-2010 years to 2099 were held at the 2010 value of US$609 million annual net losses (DARA, 2012).

Table 42: Global CO_2 emissions by decade (Million tonnes)

	1960s	1970s	1980s	1990s	2000s
Decade emissions Mt	117.75	176.50	194.50	231.50	279.75

The approach assumes that each tonne of CO_2 has equal responsibility for damage between 2010 and 2099, and also assumes a CO_2 half life of 30 years and a maximum lifetime impact of 100 years (up to 2099). Though in fact, whilst about 50% of a CO_2 increase will be removed from the atmosphere within 30 years, and a further 30% will be removed within a few centuries; the remaining 20% may stay in the atmosphere for many thousands of years (IPCC, 2007a).

A discount rate of 1% for the years 2010-2099 was adopted. The rationale for using a 1% rate is as follows. Adopting Stern's (2007) rate of time preference (0.1%) and an average GDP per capita growth rate of 1.3% gives a discount rate value of 1.4%, as used by Stern. However, whilst the damages extend beyond Argentina, all of the various discounted values in this report are from the perspective of Argentina. The historic growth rate in Argentina between 1960-2010 was lower, at around 1.1% (World Bank, 2014a). It is widely assumed that developing country growth rates will be higher over this century. However, at the same time, the future growth rate in developing countries is expected to be negatively affected by climate change impacts, by up to 0.6% per year from 2009-2099 (Dell et al., 2008). Therefore, it is assumed that whilst GDP per capita would increase at a faster rate over coming decades in Argentina in the absence of climate change impacts, the net effect of climate impacts is to reduce it to 0.9% per annum. By including the 0.1% rate of time preference, this results in an overall discount rate of 1%.

[59] Thus the 1960 impact of all carbon emissions that year is US$0.006 million (US$6,000) globally.

Using this rate the rounded average (mean) decadal damages and SCC per tonne of CO_2 is estimated (in 2014 prices) to be as follows (Table 43):

Table 43: SCC calculations using alternative approach (which use a 1% discount rate)

	1960s	1970s	1980s	1990s	2000s
In-decade damages US$ million	0.2	1.2	7.2	54.0	563.4
30 year damages US$ million	32.6	341.4	1325.2	2685.0	4278.6
Damages >30 yrs (to 2099) US$ million	1992.6	3244.9	3501.3	6127.4	6654.9
Damages US$ per tonne CO2	17	20	25	38	40

These SCC values (using a 1% discount rate) are lower each decade than the US EPA 2.5% discount rate figures projected backwards[60]. In this study, the more conservative lower carbon values calculated above using the alternative approach are adopted, along with the (lower value) EPA 5% values (as in Figure 44). However, there are arguments for why a higher SCC value should be adopted not only over coming decades but also for historic emissions. Whilst the two sets of carbon values present a range, the higher figures used should not be considered as the likely upper-bound of the SCC (e.g. see Stern (2007), where the SCC in 2006 is estimated to be US$310 per tonne of carbon, i.e. over double the average carbon cost of the alternative approach in that decade). It is hoped that over the coming years an improved estimate of the historical costs of CO_2 emissions will be available. Until then some variation on the above may be the best option for use in similar studies.

The average (mean) value (up-rated by CPI to 2014 values) for each decade based on the US EPA figures with a 5% discount rate, is presented in Table 44, firstly for CO_2, which is then converted into a carbon value. The alternative approach values are also presented.

Table 44: Average (mean) SCC values (in 2014 prices) for each decade 1960s – 2000s

US EPA (5%)

US$ per tonne of CO2	
1960s	0.96
1970s	1.61
1980s	2.68
1990s	4.48
2000s	7.49
US$ per tonne of Carbon	
1960s	3.53
1970s	5.90
1980s	9.85
1990s	16.46
2000s	27.49

Alternative (1%)

US$ per tonne of CO2	
1960s	17.20
1970s	20.33
1980s	24.85
1990s	38.30
2000s	40.49
US$ per tonne of Carbon	
1960s	63.12
1970s	74.61
1980s	91.20
1990s	140.56
2000s	148.60

[60] The alternative 1% rate gives equivalent values to a US EPA 3% discount rate.

Appendix C: Spatial analysis methodology

Why the HYDE data is used

The analyses presented in this report are based on historical land cover data from Meiyappan and Jain (2012). Whilst major caveats of this dataset include the relatively coarse spatial resolution (50x50km) and the inherent uncertainties associated with the spatial modelling techniques employed and estimates of historical land-use used to generate this dataset (for more information on the data description please see the data documentation for Meiyappan and Jain (2012), available at http://climate.atmos.uiuc.edu/ISAM_Landuse/land-cover_doc_c20130831.pdf), this dataset was best suited for the purposes of this study. In order to be able to analyse land-use change across a 50 year period, the datasets for each decade needed to be comparable in terms of the land use categories they employ. This type of study could be repeated using better, national level data as available, perhaps with a focus on only more recent decades for which national data is available. For 2000-2010, a comparison in terms of relative amounts of "forest loss" has been made with higher resolution (30m) global data on forest cover change by Hansen et al. (2013), in order to show the potential differences between two products for the recent decade.

Changes in land cover per decade 1960 – 2010

Historical land cover data from Meiyappan and Jain (2012) was used as the input data for this analysis. IDRISI Selva 17.0 Land Use Change Modeller (LCM) was used to analyse the changes in land cover for each decade, based on data for the year at the beginning and end of each decade. IDRISI Selva LCM analyses changes in land cover on a pixel by pixel basis. For the purpose of this study, forest cover was classed as land cover categories "Tropical Evergreen Broadleaf Forest"; "Tropical Deciduous Broadleaf Forest"; and "Shrubland" in the input dataset. The loss/gain in each of these categories is presented as a series of maps (Figure 8-12). For areas of forest loss, the new land-use is specified by LCM.

Reducing sedimentation through limiting soil erosion

i. Generation of a soil erosion risk layer:

The relative potential "risk" of an area in terms of soil erosion has been evaluated as a function of slope and rainfall. This method uses an overlay approach, where annual average precipitation per cell (split into 4 classes using a quantile classification) was combined with data generated for slope (split into 4 classes using a quantile classification and generated based on Digital Elevation Model data). Since there are 4 classes for both slope and precipitation, the resulting output has a maximum value of 8 and a minimum value of 2. The classes represent a low to high potential for erosion risk. Higher values represent higher erosion impact in the absence or degradation of forests. No weighting is used in this simplified approach, the relative importance of each input factor is the same. Obviously, more detailed analysis could also include data on soil type.

ii. Generation of upstream catchments of dams:

A 3-arc second (90m) resolution void-filled Digital Elevation Model (DEM) from HydroSHEDS (Lehner et al. 2008a) was used to generate flow direction, flow accumulation and a stream order network for Argentina. Using these layers and dam data (Lehner et al. 2008b), upstream catchments of dams were created, and validated using the stream order network. The calculations presented in Table 29 have been based on the pixels of forest loss falling within dam catchments. This methodology has accounted for the year of dam construction in the generation of these statistics.

Impacts on water flows: floods and droughts

Using data from FAO AQUASTAT on major hydrological basins (Derived from HydroSHEDS), the amount of forest loss per decade falling within each major basin was calculated. The forest loss data for each of the decades was clipped to each of the major basins and the areas were calculated.

Carbon retention / sequestration

The average amount of biomass carbon (tonnes/ha) was calculated using biomass carbon data by Ruesch and Gibbs (2008). This is a global map of biomass carbon stored in above and belowground living vegetation for the year 2000. Using zonal statistics, the historical land cover data for this year from Meiyappan and Jain (2012) was then used as zonal input data to generate statistics. For each land cover type, the average amount of biomass carbon (tonnes/ha) was calculated, and the figures for each land use type were used as a basis for the calculations on carbon presented in this report.

Harvested NWFPs / sustainably harvested wood

Using point data for towns in Argentina, the loss of potential NWFP harvest area for these population centres was calculated based on a 6km accessibility buffer, applied around villages and towns. The type and amount (km^2) of forest for each decade falling within this buffer was calculated by clipping forest to the buffered areas and calculating the proportion of forest falling within these.

Biodiversity and landscapes as a tourism resource

This analysis was based on mammal, bird, amphibian and reptile species ranges classified as threat status 'Critically Endangered', 'Endangered', and 'Vulnerable' by the IUCN Red List of Threatened Species (2014) Version 2014.2. A 10 km^2 hexagon grid covering Argentina was generated using Jenness Enterprises repeating shapes tool in ArcGIS 10.2. Hawths Analysis “enumerate intersecting features” tools were then used to generate species richness by calculating the number of species ranges intersecting with each hexagon. Hexagons were then shaded by species number. Areas with >8 overlapping threatened species ranges were considered in this study to be of high threatened species richness. Forest loss for each decade falling within Key Biodiversity Areas (KBAs), Protected Areas, and areas of high threatened species richness was calculated by clipping the forest loss for each decade to these areas. Note that a number of these areas are overlapping; this was taken into account within the statistical calculations.

Appendix D: Table of results

The full range of results from the Argentina case-study is presented below:

Account of the net gains from deforestation between 1960 and 2010

Gains from deforestation:	US$ (2010) billion
Value of crop production[1]	191.4 - 356.3
Value of livestock production[1]	33.1 - 75.7
Value of the felled wood[2]	0.6 - 1.2
Total	225.1 - 433.2

Losses from deforestation:	US$ (2010) billion
Climate change impacts (SCC)[1,3]	10.6 - 85.8
Water regulation[1]	6.5 - 307.5
Sustainably harvested wood products[1]	10.1 - 34.3
Non-wood forest products[1]	6.0 - 10.1
Other (international tourism, sedimentation of reservoirs, disease)[1]	0.3 - 1.0
Unvalued (e.g. pollination, recreation/cultural, genetic)[4]	?
Total	33.5 - 438.7(+?)

The net result ranges from: a 15% reduction in the gains, to an overall loss[5].

Note: range of values taken from analysis, reflecting different discount rates and/or underlying assumptions (maximum and minimum values, as noted at various points in the analysis, have been used to produce this range).

1 - past and future (up to year 2099)
2 - at time of felling (range reflects assumed wood utilization rate)
3 - SCC is the social cost of carbon (i.e. the economic impact of CO_2 emissions)
4- based on other studies, together these could be worth in excess of US$100 billion
5 - loss would mean that the deforestation that occurred between 1960 and 2010 was 'uneconomic'

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