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Prospective Environmental Impacts of Selected Low-Carbon Electricity Technologies

Doctoral Thesis

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Faculty of Engineering Science and Technology
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Preface

This thesis was written in partial fulfilment of the degree of philosophiae doctor at the Department of Energy and Process Engineering of the Norwegian University of Science & Technology (NTNU). The work was carried out at the Industrial Ecology Programme at NTNU during a three-year period from June 2012 to September 2015 and included coursework totaling 30 European Credits (one academic semester). Two years of the research period was funded by the Research Council of Norway (contract no. 206998) and the remainder by the Department of Energy and Process Engineering at NTNU.

Evert A. Bouman

Trondheim, November 2015

Abstract

As one of the main contributors to greenhouse gas (GHG) emissions, the global electricity production sector faces the challenge of mitigating its emissions by transitioning towards cleaner production technologies. In light of this transition, it has been shown that even though renewable energy technologies have clear benefits over fossil generation technologies, there are trade-offs from an environmental and material perspective (Hertwich et al. 2015; Singh et al. 2015). Within the broader scope of sustainable development, care must be taken in climate change mitigation to avoid problem-shifting between environmental impacts.

The aim of this thesis is to shed light on the prospective environmental impacts of low-carbon electricity production technologies. Life Cycle Assessment (LCA) was chosen as assessment method for its comprehensive scope. It is detailed enough to estimate prospective impacts with a level of detail that is typically not available for other types of environmental assessment methods. Two key challenges associated with the LCA of electricity technologies were identified: input data variability and uncertainty, and the challenge of matching electricity supply and demand due to intermittent solar and wind resources. It is of importance to take these challenges into account in the environmental assessment of electricity technologies.

Against this background, four papers are presented in this thesis. The first paper discusses the influence of fugitive methane emissions on the life cycle GHG emissions of fossil fuel based electricity generation. The other three papers investigate more closely the environmental impacts of electricity technologies, while taking into account the variability and market dispatch of supply from renewable resources. *Paper II* aims to quantify the additional environmental impacts of extending an offshore wind farm with compressed air energy storage (CAES) for balancing purposes. The influence of economic electricity dispatch on capacity factor assumptions and environmental impacts is studied in *Paper III*. The prospective environmental impacts of high-renewable electricity production regimes, including intermittency and economic dispatch, are discussed in the final paper.

It is shown in *Paper I* that GHG emissions from fossil fuel power generation vary more widely than commonly acknowledged as a result of large variability in fugitive methane emissions. Where CO_2 capture and storage (CCS) reduces the GHG emissions at power plants, it increases the upstream fugitive emissions per unit generation. The high variability in results points to a need for more measured data to reduce the uncertainty in the dataset, as well as a potential mitigation opportunity by capturing methane during fossil fuel extraction.

It is shown in *Paper II* that the additional environmental impacts related to balancing offshore wind power with CAES are limited and combined impacts are well below average grid impacts. Both conventional and adiabatic CAES are investigated. The majority of environmental impacts can be attributed to either the combustion of natural gas in conventional compressed air systems, or the thermal energy storage in adiabatic systems.

It is shown in *Paper III* that the utilization of electricity technologies is determined by the economic dispatch, and is of influence on the capacity factor estimation of individual technologies. The sensitivity of impact assessment for the capacity factor estimation is large for infrastructure intensive technologies and can vary across different impact categories.

It is shown in *Paper IV* that increasing the share of renewable electricity in the mix, decreases impacts significantly. Impact indicators are aggregated into a single impact according to four distinct weighting methods. Large variation in impact is observed for scenarios with comparable levels of renewable electricity production, but different shares of individual technologies. Specific renewable technology targets could inform the setting of renewable energy targets motivated by impact reduction.

The thesis as a whole shows that, while the impacts of low-carbon electricity cannot be underestimated, there is significant environmental improvement potential related to the transition to a low-carbon electricity system. The additional environmental pressures associated with balancing electricity supply and demand appear to be relatively limited. The impact reductions achieved by additional low-carbon capacity more than outweigh the potential adverse effects related to its construction.

Sammendrag

Den globale elektrisitetsproduksjonssektoren er en av de viktigste bidragsyterne til utslipp av klimagasser (GHG) og står foran utfordringen ved å gå over til renere produksjonsteknologier for å redusere disse utslippene. Selv om fornybare energiteknologier har klare fordeler sammenlignet med fossile teknologier, finnes det avveininger fra et miljø- og materialperspektiv (Hertwich et al 2015;. Singh et al 2015). Innenfor den bredere rammen av bærekraftig utvikling, må man passe på å gjennomføre klimatiltak som unngår forskyvning av problemene mellom ulike miljøpåvirkninger.

Målet med denne avhandlingen er å kaste lys over de potensielle miljøkonsekvensene av elektrisitetsproduksjonsteknologier med lave utslipp av klimagasser. Livsyklusvurdering (LCA) er valgt som vurderingsmetode på grunn av sitt omfattende omfang. Det er spesifikt nok til å anslå potensielle konsekvenser med et detaljnivå som tradisjonelt ikke er tilgjengelig for andre typer miljøvurderingsmetoder. To viktige utfordringer knyttet til LCA av elektrisitetsteknologier ble identifisert: variabilitet og usikkerhet fra inndata, og utfordringen med matchende elektrisitetsforsyning og -etterspørsel på grunn av uregelmessige sol- og vindressurser. Det er av betydning å ta disse utfordringene i betraktning i miljøvurderinger av elektrisitetsteknologier.

Mot denne bakgrunnen er fire artikler presentert i denne avhandlingen. Den første artikkelen diskuterer innflytelsen av metanlekkasje på klimagassutslippene forbundet med fossilbasert kraftproduksjon. De tre andre artiklene undersøker nærmere de miljømessige konsekvensene av kraftteknologier, med samtidig hensyn til variasjon av produksjon fra fornybare ressurser og økonomisk driftsplanlegging av levering. *Paper II* tar sikte på å kvantifisere de ekstra miljømessige konsekvensene av å utvide en vindpark nær kysten med trykkluftsbasert energilagring (CAES) for balanseringsformål. Påvirkningen av økonomisk driftsplanlegging på kapasitetsfaktorforutsetninger og miljøkonsekvenser er studert i *Paper III.* De potensielle miljømessige konsekvensene av høy-fornybare regimer, inkludert variabilitet av levering og driftsplanlegging, er omtalt i den siste artikkelen.

Det er vist i *Paper I* at klimagassutslipp fra fossil kraftproduksjon varierer mer enn tradisjonelt antatt som et resultat av stor variasjon i metanutslipp. CO_2-fangst og lagring (CCS) reduserer klimagassutslippene på kraftverk, men øker oppstrøms behov for brennstoff og derfor også lekkasje per enhet produksjon. Den høye variasjonen i resultatene peker på et behov for flere målepunkter for å redusere usikkerheten i datasettet, samt en potensiell mulighet for å redusere klimagassutslipp ved å fange metan ved utvinning av fossile brensler.

Det er vist i *Paper II* at økningen i miljøeffekter knyttet til å balansere offshore vindkraft med CAES er begrenset og at totaleffektene er betydelig lavere enn gjennomsnittet for kraftproduksjon. Både konvensjonell og adiabatisk CAES blir undersøkt. De fleste av miljøeffektene kan tilskrives enten forbrenning av naturgass i konvensjonelle trykkluftsystemer, eller lagring av termisk energi i adiabatiske systemer.

Det er vist i *Paper III* at utnyttelsen av kraftproduksjonsteknologier er bestemt av økonomisk driftsplanlegging, og har innflytelse på estimering av kapasitetsfaktor for individuelle teknologier. Miljøvurderinger av infrastruktur-intensive teknologier er følsomme for estimater av kapasitetsfaktor og kan variere mellom ulike påvirkningskategorier.

Det er vist i *Paper IV* at å øke andelen av fornybar elektrisitet i miksen, reduserer miljøeffektene betydelig. Miljøeffekter er samlet i henhold til fire forskjellige vektingsmetoder. Stor variasjon i effektene er observert for scenarier med sammenlignbare nivåer av fornybar elektrisitetsproduksjon, men ulike andeler av de ulike teknologiene. Spesifikke mål for individuelle fornybare teknologier kan forbedre utformingen av fornybare energimål motivert av miljøhensyn.

Avhandlingen som helhet viser at mens konsekvensene av elektrisitet med lave CO_2 utslipp ikke bør undervurderes, er det betydelig miljøforbedringspotensial knyttet til overgangen til et kraftsystem basert på lavutslippsteknologi. De ekstra miljøbelastninger i forbindelse med behovet for å balansere kraftsystemet synes å være begrenset. Gevinstene oppnådd ved ny lavutslippskapasitet mer enn oppveier de potensielle negative effektene knyttet til oppbygging av den tilhørende infrastruktur.

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Knowingly and unknowingly, my colleagues at the Industrial Ecology Programme (IndEcol) contributed in the writing of this thesis. First and foremost by creating a wonderful research atmosphere in which creative thinking is stimulated and ideas are nourished. In addition, I am grateful to have been working in a great social atmosphere; one that extends beyond the confinements of office space and hours. Special thanks go to: Thomas Gibon, Bhawna Singh, Linda Ellingsen, Guillaume Majeau-Bettez, Tuva Grytli, Moana Simas, Konstantin Stadler, Stefan Pauliuk, Helen Hamilton, Carine Lausselet, Franciska Steinhoff, Dan Moran, Richard Wood, Francesco Cherubini, Anders Strømman, Helge Brattebø, Daniel Müller, and all others at IndEcol.

For their valuable contributions to the papers, I owe thanks to: Andrea Ramirez, Martha Marie Øberg, Christian Skar, and of course Edgar Hertwich.

I would like to thank Kjartan Steen-Olsen and Anders Arvesen for their help in writing the Norwegian summary of this work.

I would also like to take this opportunity to thank my mother and sister for supporting my move to Trondheim, even though my departure has at times been hard for them and we do not see each other as often as we would like.

Above all, I acknowledge the unconditional love and support I receive from my wife Rita. She has been instrumental in the finishing of this work and I am greatly indebted to her.

Evert A. Bouman

Trondheim, November 2015

Table of Contents

List of articles

This thesis is based on the articles written during my period as a PhD candidate over the past three years. All articles have been co-authored and four articles form the main backbone of this work. I have been the lead author of these works and I listed them below as primary publications. The articles are in integral form appended to this thesis as appendices A-D. For the articles that are not yet published (B-D), I have also included the Supporting Information. In addition to the primary publications, I have appended a secondary publication (E) for which it was my pleasure to be a co-author. I believe this secondary work contains relevant background information and I draw heavily upon it. I further list here the other projects I have been involved in during the course of my PhD.

Primary publications

Paper I

Bouman, E. A., A. Ramirez, and E. G. Hertwich. 2015a. Multiregional environmental comparison of fossil fuel power generation—Assessment of the contribution of fugitive emissions from conventional and unconventional fossil resources. *International Journal of Greenhouse Gas Control* 33(0): 1-9.

Author contribution: Research co-design, data collection, modelling, analysis, and writing.

Paper II

Bouman, E. A., M. M. Øberg, and E. G. Hertwich. 2015b. Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES). *Under review with Energy*

Author contribution: Research co-design, data collection, modelling, analysis, and writing.

Paper III

Bouman, E. A., C. Skar, and E. G. Hertwich. 2015c. LCA of electricity technologies using capacity factors dependent on economic dispatch. *Submitted to Environmental Research Letters*

Author contribution: Research idea and design, data collection, modelling, analysis, and writing

Paper IV

Bouman, E. A., C. Skar, and E. Hertwich. 2015d. Specific renewable energy technology targets can reduce life cycle impacts of electricity generation. *Submitted to Environmental Science & Technology*

Author contribution: Research idea and design, data collection, modelling, analysis, and writing.

Secondary publications

Supporting Paper I

Hertwich, E. G., T. Gibon, E. A. Bouman, A. Arvesen, S. Suh, G. A. Heath, J. D. Bergesen, A. Ramirez, M. I. Vega, and L. Shi. 2015. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. *Proceedings of the National Academy of Sciences* 112(20): 6277-6282.

Other work

Singh, B., E. A. Bouman, A. H. Strømman, and E. G. Hertwich. 2015. Material use for electricity generation with carbon dioxide capture and storage: Extending life cycle analysis indices for material accounting. Resources, Conservation and Recycling 100(0): 49-57.

Majeau-Bettez, G., S. Pauliuk, R. Wood, E. A. Bouman, and A. H. Strømman. 2015. Balance issues in input–output analysis: A comment on physical inhomogeneity, aggregation bias, and coproduction, *under review*

A. Ramirez et al. C*hapter 3: Fossil fuels,* in: Hertwich, E. G., J. Aloisi de Larderel, J. Bergesen, S. Suh, S. Li, and T. Gibon. 2015. Green Power Choices: The benefits, risks and trade-offs of low carbon technologies for electricity production. United Nations Environment Programme, *in press*

Hertwich, E.G., J. Weinzettel, E. A. Bouman, T. Gibon, A. Arvesen, and J. Knapek. *Chapter 9: Matching Supply and Demand: Grid and Storage,* in: Hertwich, E. G., J. Aloisi de Larderel, J. Bergesen, S. Suh, S. Li, and T. Gibon. 2015. Green Power Choices: The benefits, risks and trade-offs of low carbon technologies for electricity production. United Nations Environment Programme, *in press*

List of Figures

1 Introduction

As concern over the environment grows and the evidence of human induced climatic change becomes more apparent, a trend is developing to incorporate potential environmental impacts into the policy making process concerning climate change mitigation[1] technologies and measures. Examples are increasing fuel efficiency standards for cars, power generation using fossil fuels with carbon dioxide capture and storage (CCS), increased recycling schemes, and the banning of heavily polluting and toxic chemicals in the chemical process industry.

The transformation of electricity systems is expected to play a major role in transferring to a low-carbon society. The global electricity sector contributes annually with more than 12 Gt of CO_2 to climatic change and mitigation opportunities are significant (Bruckner et al. 2014). In this thesis, I focus on the prospective environmental impacts of low-carbon electricity technologies. A low-carbon electricity technology produces power with lower than average carbon dioxide emissions. The utility of these technologies in the mitigation of climate change is beyond doubt. However, this does not exclude the possibility of the existence of other adverse environmental impacts. The resources and processes required to build and operate low-carbon electricity technologies differ significantly from those of 'traditional' power stations. As a result, the environmental impact profiles of these technologies can also be expected to differ.

The existing and future energy infrastructure constitutes and defines largely the human commitment to a global warming pathway. Due to the large investment costs associated with electricity infrastructure, a technological lock-in effect is created (Unruh 2000). The possibility of path-dependency and lock-ins of climate mitigation strategies is highlighted by the Intergovernmental Panel on Climate Change (IPCC), as

[1] Mitigation is a reduction in the unpleasantness, seriousness, or painfulness of something. In the context of climate change, mitigation refers to a human intervention to reduce the sources or enhance the sinks of greenhouse gases (IPCC 2014).

some climate responses can have adverse side-effects and generate risks in addition to potential co-benefits[2] (Fleurbaey et al. 2014). These side-effects can occur within economic, social, and environmental domains. For example, increased use of CCS replacing conventional coal raises the requirements for physical capital in the fossil industry (economic), has a health impact via the risk of CO_2-leakage and upstream supply-chain activities (social), and increases water-use and ecosystem impact (environmental). Valid concerns related to the large-scale deployment of renewable technologies are the incurrence of extra costs related to intermittency issues (economic), threat of displacement of human settlements in the case of large hydro projects (social), and increased water use and habitat impacts (Clarke et al. 2014). Also, wind turbines may cause visual and noise pollution, the large-scale manufacture of PV technology may lead to significant resource depletion, and hydro power and biomass can cause land use problems (Masanet et al. 2013; Fthenakis and Kim 2009).

It has to be kept in mind that climate change is only one of the challenges that are presented to society in the coming decades. Other identified challenges are, for example, access to clean and reliable energy, limiting air pollution, health damages, water impacts, and biodiversity loss (Clarke et al. 2014). Rockström et al. (2009) have proposed a framework of 'planetary boundaries', which define the safe operating space for humanity with respect to the Earth system. Crossing these boundaries could generate unacceptable environmental change. Rockström et al. (2009) report an exceedance of boundaries for climate change, rate of biodiversity loss, and the interference with the nitrogen cycle. In addition, boundaries for global freshwater use, land-use change, ocean acidification, and interference with the global phosphorous cycle may be quickly approached.

In summary, within the broader scope of sustainable development, care must be taken in climate change mitigation to avoid problem-shifting between impacts. As the electricity sector contributes significantly to climate change, a comprehensive

[2] Co-benefits (ancillary benefits) are the positive effects that a policy or measure aimed at one objective might have on other objectives (IPCC 2014).

overview of a wide range of prospective environmental impacts of low-carbon electricity technologies is paramount.

In the following section of this Chapter, I will introduce what are, in my opinion, the key elements relating to climate change, the expected changes in the electricity system, and the status of the (life cycle) environmental assessment of electricity generation technologies. The section is by no means intended to give a comprehensive review and discussion of these three topics. Rather, it is meant to illustrate the background relating to this thesis and the context of the cases described in the appended papers. I continue the Chapter with a short summary of the emergence of the field of Industrial Ecology and the development of common tools and methods used by industrial ecologists. The research questions and motivation are described in further detail in Section 1.3, and I conclude this Chapter with a description of the structure of the remainder of this thesis.

1.1 Background

1.1.1 Anthropogenic climate change

The concept of atmospheric warming by means of greenhouse gases is not new. Svante Arrhenius estimated CO_2 induced warming already in 1896 (Arrhenius 1896). In the past decades, the temperature of the atmosphere and oceans has risen at an unprecedented rate, sea levels have risen, and the concentrations of greenhouse gases (GHGs) in the atmosphere have increased. GHGs contribute to positive radiative forcing[3] ('global warming') and the largest contribution to total radiative forcing is caused by the gradual build-up of CO_2 since the beginning of the industrial revolution (IPCC 2013). Annually, around 14.7 Gt CO_2 is added to the atmosphere due to human activity (averaged over the time period 2000-2009). Hence, we speak of the term anthropogenic climate change: climate change of human origin. Actual emissions from human activity are higher, but a large part of emissions is taken up and stored by the oceans and terrestrial ecosystems (Ciais et al. 2013).

[3] Difference of insolation absorbed by the Earth and energy radiated back to space.

Figure 1 shows the change in global annual mean surface temperature relative to 1986-2005 as projected by the IPCC. Indicated is the temperature increase corresponding to high and low GHG emissions scenarios, so-called Representative Concentration Pathways (RCP)[4].

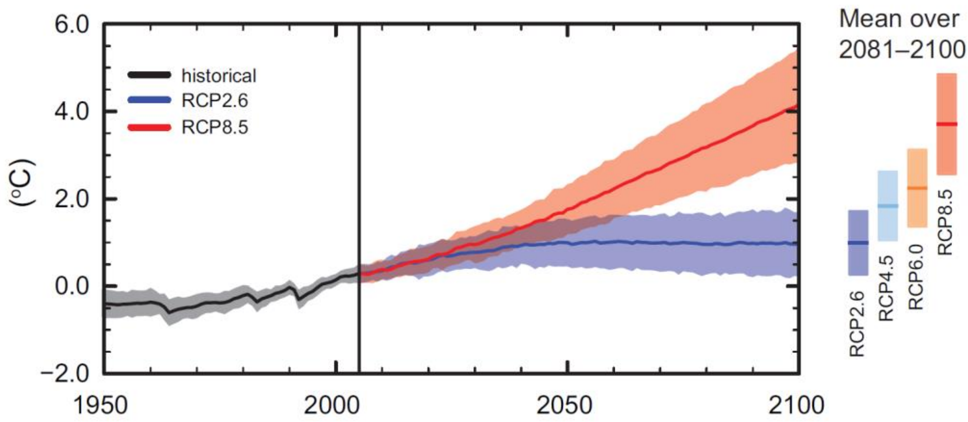

Figure 1: Change in global annual mean surface temperature relative to 1986-2005. Time series of projections and a measure of uncertainty are shown for the scenarios RCP2.6 (blue) and RCP8.5 (red). Black (grey shading) is the modelled historical evolution. The colored vertical bars indicate the mean and uncertainty averaged over the time period 2081-2100 for all RCP scenarios. Time series were results of the Coupled Model Intercomparison Project, CMIP5 (Taylor et al. 2012) and do not result from a single climate model. Source: adapted from IPCC (2013).

Uncurbed GHG emissions lead to unparalleled increases in global surface temperature. A large amount of nations have expressed a wish to limit global warming to 2 °C above pre-industrial level. Thus, a significant decrease in GHG emissions currently resulting from human activity is to be achieved, and eventually emissions will have to cease entirely. The upper limit of emissions is captured in the notion of a carbon budget, the

[4] The Representative Concentration Pathways (RCPs) are a set of four pathways as a basis for long-term and near term climate modeling experiments and are designated RCP8.5, RCP6, RCP4.5, and RCP2.6. The values indicate the radiative forcing target level in W/m^2 in 2100. The forcing estimates are based on the forcing of GHGs and other forcing agents. The RCPs are generated by four different integrated assessment models, meant as internally consistent pathways, and are, as a set, representative of a larger set of emissions scenarios currently available in the scientific literature (van Vuuren et al. 2011).

amount of carbon dioxide that can be emitted without overshooting the 2 °C threshold. Meinshausen et al. (2009) calculate a limit of 1000 Gt CO_2 in the period 2000-2050, which yields a 25% probability of warming exceeding 2 °C, and a limit of 1440 Gt CO_2, which yields a 50% probability. Davis et al. (2010) estimate that the commitment of existing infrastructure to future emissions and global warming is, cumulatively, 496 (282-701) Gt CO_2 from combustion of fossil fuels between 2010 and 2060. McGlade and Ekins (2015) estimate the amount of fossil fuel reserves that are unburnable before 2050 to be 33%, 49% and 82% for respectively oil, gas, and coal in a 2 °C scenario with a widespread deployment of CCS technology from 2025 onwards.

A breakdown of total annual anthropogenic GHG emissions in 2010 by economic sector, expressed in CO_2-equivalent, is shown in Figure 2. The most significant source of emissions is the energy sector (including heat and electricity production) followed by emissions from agriculture, forestry and other land use and direct emissions from industry.

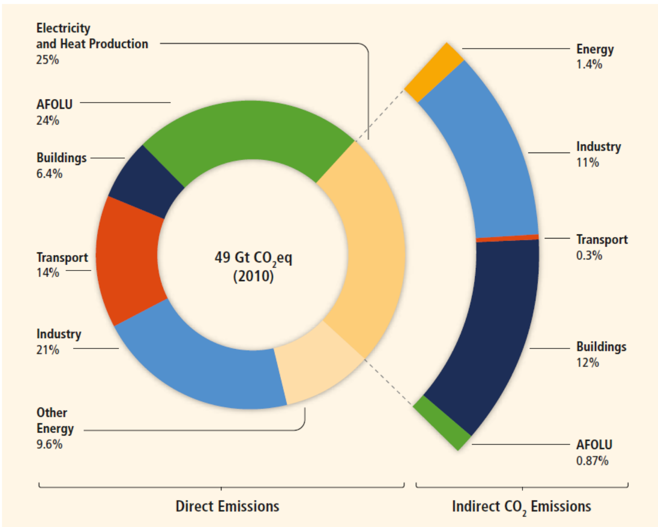

Figure 2: Total anthropogenic greenhouse gas emissions estimate for 2010 by economic sector. Shown are direct emission shares (% of total anthropogenic GHG emissions) for five major economic sectors. The pull-out shows how indirect CO_2 emissions shares can be attributed to sectors of final energy use. AFOLU = Agriculture, Forestry and Other Land Use. Source: Edenhofer et al. (2014).

The heat and electricity sector is likely to face a significant transformation if the 2 °C target is to be achieved. In addition to causing a substantial portion of GHG emissions, emissions mainly occur at point sources, i.e. power plants. These sources have the potential of being retro-fitted with emissions control equipment, or being replaced with cleaner technology and are therefore a natural mitigation option. In the next subsection, we briefly highlight some scenarios with respect to this transition.

1.1.2 Transition of the global electricity system

The executive summary of the most recent Energy Technology Perspectives report from the International Energy Agency (IEA) starts as follows: *"Energy technology innovation is central to meeting climate mitigation goals while also supporting economic and energy security objectives."* (IEA 2015)

The IEA uses three main scenarios in its assessments: 6DS, 4DS, and 2DS. The 6DS scenario assumes no GHG mitigation efforts beyond those policy measures already implemented and projects an increase in the emissions rate to 56 Gt CO_2 per year by 2050. The 4DS scenario projects an emissions rate of 41 Gt CO_2 per year and assumes that climate and energy policies that are being planned or under discussion are implemented. The most ambitious of the IEA scenarios is the 2DS scenario, which describes a pathway with a 50% chance of limiting global mean temperature increase to 2 °C. Its pathway consists of reaching an annual emissions level of 14 Gt CO_2 by 2050 (IEA 2015).

In the previous subsection the annual GHG emissions from the heat and electricity sector are shown to be around 12 Gt CO_2-eq, well within the 14 Gt CO_2 required for the 2DS scenario. However, as both the global population, and its affluence, are expected to increase significantly in the coming decades, an increase in electricity production can be expected that lies between 75% (2DS) and 110% (6DS) in a period of 40 years. Thus, mitigation efforts must focus on increases in efficiency, decreases of electricity use, and a maximum utilization of electricity from renewable sources. Figure 3 shows the electricity generation pathways for different generation technologies in both the 6DS and 2DS scenario. In the 2DS scenario, coal and natural gas are expected to decrease production volumes significantly, and an increased penetration of renewable electricity technology and nuclear power is predicted. Little is known about the environmental implications of globally increased use of low-carbon electricity technologies. In the following subsection, I summarize the status regarding the environmental assessment of low-carbon electricity technologies.

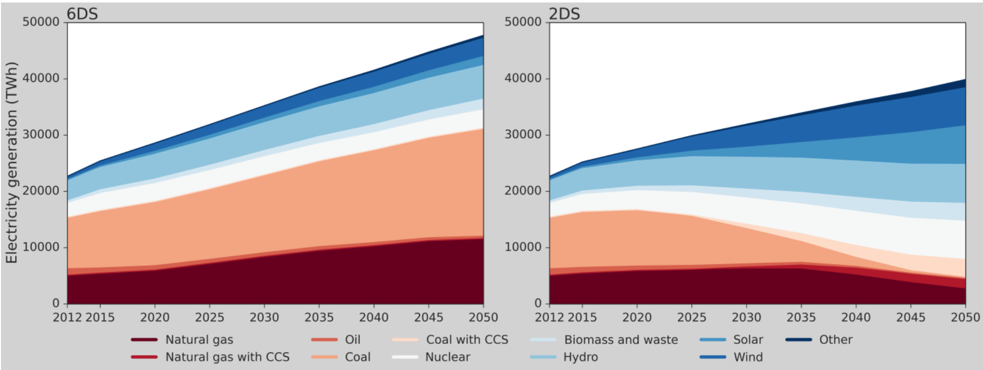

Figure 3: Electricity generation for different types of technology for the 6DS and 2DS scenario. Source: IEA (2015).

1.1.3 Environmental assessment of electricity generation technologies

As a general term, environmental assessment refers to the activity and process of analyzing, evaluating, and judging (impacts on) the environment. Environmental assessment might also refer to the formal procedure that ensures that the environmental implications of decisions are taken into account before the decisions are made (EC 2015). A large number of tools and methods are available for assessing the environmental impacts of technologies, differing in scope, detail, and objective. They can take a bottom-up or top-down perspective, range from site-specific to global coverage, and focus on one, or multiple, impacts such as GHG emissions, land area, and material use (Finnveden and Moberg 2005).

In the context of this thesis, I will describe environmental impacts mainly based on results obtained with Life Cycle Assessment (LCA). LCA aims to assess the environmental impacts associated with product systems over their entire life cycle. It captures both direct and indirect impacts, for a wide range of environmental impact categories. This makes it an ideal starting point for the assessment of electricity technologies. LCA is a bottom-up method that relies on previously established databases containing detailed process and product information, allowing to estimate prospective impacts with a level of detail that is typically not available for other types of environmental assessment methods. For example, Integrated Assessment Models (IAMs), which are traditionally used to assess large-scale scenarios development pathways from a top-down perspective, are excellent tools to assess interdependency between sectors and systems (Bauer 2015). However, they generally focus on GHG emissions levels and do not incorporate other environmental impacts to the extent and detail available for LCA studies (Masanet et al. 2013). Buonocore et al. (2015) link an economic dispatch model to an atmospheric chemistry, fate, and transport model, to estimate public health effect of displacing conventional fossil plants with renewable technology. This model, however, relies on standard emissions factors, and does not capture in detail the division between direct and indirect environmental impacts.

LCA is at the same time not too detailed. For example, Environmental Impact Assessment (EIA) is an established tool for assessing the environmental impacts of projects. It is generally applied to specific sites and projects, and often used to determine alternative locations for projects (Finnveden and Moberg 2005). The level of detail required, make it a tool that is ideal for assessing individual projects that are in final stages of design, approaching implementation, but make it less suitable for the kind of prospective analysis required within the scope of this thesis.

There are numerous examples of life cycle studies of individual electricity production technologies. Turconi et al. (2013) review a total of 167 case studies covered in 33 LCA publications published in the 15 years prior to the appearance of their article. They report a large variability of existing LCA results regarding the environmental consequences of electricity technologies. Results were found to be dependent on methodological aspects of LCA (e.g. the definition of the functional unit, the method employed, and allocation principles), as well as technological aspects pertaining to the case studies, such as the electricity mix applied in the model. It is the inconsistency among studies that formed the motivation for the meta-analysis performed in the LCA Harmonization project of the National Renewable Energy Laboratory (NREL) (Heath and Mann 2012). Meta-analysis, or harmonization, adjusts previously published estimates on the basis of a consistent set of methods and assumptions. The goal is to understand the range of results, reduce the variability in published results, and clarify the central tendency of published estimates (Heath and Mann 2012). Examples of harmonized LCA results are available for the following technologies: coal (Whitaker et al. 2012), conventional natural gas (O'Donoughue et al. 2014), shale gas (Heath et al. 2014), nuclear power (Warner and Heath 2012), wind power (Dolan and Heath 2012), crystalline silicon and thin-film PV power (Hsu et al. 2012; Kim et al. 2012), and concentrating solar power (Burkhardt et al. 2012).

Meta-analysis adjusts and decreases some of the variability across studies, but a truly comparative study compares all electricity technologies within a single LCA model, in order to ensure the use of common background processes, system boundaries, functional unit, etc. Hertwich et al. (2015) (Paper attached as Appendix E) compare 17 low-carbon electricity technologies and 4 conventional fossil technologies in such a common framework. In addition, results are calculated for nine different world regions as a means of performing regional sensitivity analysis. In Figure 4, the results of the environmental comparison of life-cycle environmental pressures per MWh electricity generation are shown. Renewable energy technologies have much lower pollution-related environmental impacts per unit generation than conventional fossil generation, but tend to require more bulk materials. Per unit generation, application of CCS technology to fossil power plants results in an increase of non-GHG impacts and material requirements, due to the inherent efficiency penalty related to the CCS process, a conclusion also reached by e.g. (Singh et al. 2011; Kleijn et al. 2011; Singh et al. 2015). The regional sensitivity, indicated by the disparity of the data points in Figure 4, seems to be largest for the bulk materials used in renewable technologies, though for some technologies we see significant disparity in GHG emissions, particulate matter formation, and land occupation. In addition to the above described results, Hertwich et al. (2015) also calculate the life cycle impacts of the IEA BLUE Map scenario (IEA 2010) (comparable to the 2DS scenario described above). I do not summarize these results here but instead refer to appendix E.

Hertwich et al. (2015) point out that the additional environmental impacts of balancing electricity supply and demand is not included in their study, as the BLUE map scenario contains a relatively modest estimation of variable wind and solar penetration. However, at higher utilization of variable renewable energy sources, both balancing, grid expansion and energy storage become of potential serious concern. In this thesis, I build on the work of Hertwich et al. (2015) to capture some of these aspects.

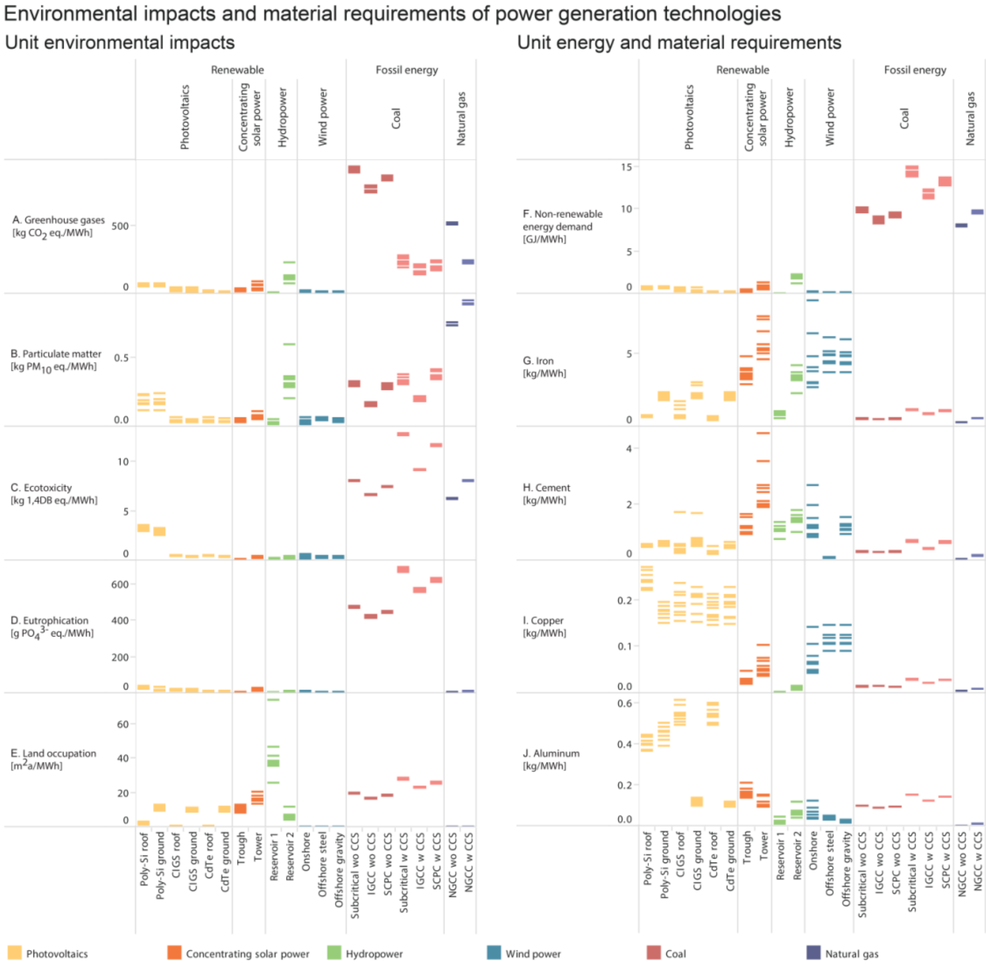

Figure 4: A comparison of life-cycle environmental pressures and resource use per unit of electricity generated by different power-generation technologies in nine world regions. The left column shows four pollution-oriented indicators: (A) Greenhouse gases, (B) particulate matter exposure, (C) freshwater ecotoxicity, and (D) freshwater eutrophication. In addition, land occupation (E) is shown. The right column indicates non-renewable primary energy demand (F) and the demand for materials (G–J). CCS, CO_2 capture and storage; CdTe, cadmium telluride; CIGS, copper indium gallium selenide; IGCC, integrated gasification combined cycle coal-fired power plant; NGCC, natural gas combined cycle power plant; offshore gravity, offshore wind power with gravity-based foundation; offshore steel, offshore wind power with steel-based foundation; reservoir 2, type of hydropower reservoir used as a higher estimate; SCPC, supercritical pulverized coal-fired power plant. Source: Hertwich et al. (2015).

1.2 Industrial Ecology and its toolbox

The emergence of the scientific field of Industrial Ecology[5] is often attributed to Frosch and Gallopoulos (1989) who describe in their seminal paper how the concepts of 'industrial ecology' and an 'industrial ecosystem' can be used to achieve an optimization in the consumption of material and energy, as a means of creating a society that is more sustainable in the face of increasing problems with waste and pollution. Industrial Ecology is usually more formally defined as: *"...the study of flows of material and energy in industrial and consumer activities, of the effects of these flows on the environment, and of the influences of economic, political, regulatory, and social factors on the flow, use, and transformation of resources."* (White 1994).

It should be noted that others have coined other definitions for Industrial Ecology (e.g. see Jelinski et al. (1992) and Graedel and Allenby (1995)) but recurring elements include the notion that within Industrial Ecology industrial systems cannot be viewed in isolation, but have to be seen in concert with surrounding systems, and that the exchanges (i.e. flows) between these systems are worth studying and could be used to as a means to decrease impacts by deliberately adjusting these exchanges. Garner and Keoleian (1995) characterize Industrial Ecology with a total of 9 attributes. The ones most relevant to this thesis I repeat here:

- a systems view of the interaction between industrial and ecological systems
- the study of material and energy flows and transformations
- an orientation toward the future
- a multidisciplinary approach

The study and description of material and energy flows and their interaction with the environment requires the collection of environmentally relevant information, which can be used in decision making, and various tools for analysis have been developed (Hond 2000). van Berkel et al. (1997) classify industrial ecology tools into four separate, but inter-related, functional types: i) inventory tools, which enable the

[5] The term Industrial Ecology was reportedly used already in the 1940s, see e.g. Renner (1947).

identification and quantification of environmental interventions ii) improvement tools, which facilitate the generation of improvement options for products, iii) prioritization tools, which provide a structured approach for evaluation and prioritization of environmental interventions, and iv) management tools, which specify procedures and routines for the development of industrial ecology projects. Examples of each of these tools are respectively: Life Cycle Inventory (LCI), Product Improvement Matrix (PIM) and Pollution Prevention Strategy (PPS), Life Cycle Assessment (LCA) and Life Cycle Cost (LCC) calculation, and Design for Environment (DfE).[6]

The development of tools in Industrial Ecology has not come to a halt. Part of the work for this thesis has been devoted to the development of tools and their application is exemplified in the studies presented in the appended papers. Using the typology of van Berkel et al. (1997) I would describe them as prioritization tools. All work in this thesis aims to provide a deeper understanding and quantification of the environmental effects related to electricity generation technologies and their interplay within an electricity system. A full overview of all methods used is given in Chapter 2.

1.3 Research questions and motivation

In the previous sections, I have illustrated some of the general problems relating to climate change, described the anticipated transition of the electricity system, and presented the status of the environmental assessment of electricity generation technologies. The above outlined issues led to the following motivation for this thesis.

Given the major role the electricity sector plays in climate change mitigation, this thesis is intended to contribute to an increased understanding of the prospective

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[6] It is interesting to note that van Berkel et al. (1997) make a distinction between LCI and LCA (described as Life Cycle Evaluation), perhaps because, at the time, LCI was already well-developed and the impact assessment component of LCA was relatively new (Klöpffer 2006). Nowadays, most would consider LCI an integral part of LCA, rather than describing them as separate tools.

environmental impacts of low-carbon electricity technologies. There are many ways of assessing impacts, some of them discussed in the sections above. LCA was chosen as main method of analysis. Life Cycle Assessment (LCA) is the ideal starting point to provide a general understanding of the potential environmental impacts of products and processes, not only during use or operation, but over the entire life cycle (Freidberg 2015). As such, it is possible to compare products and processes that provide the same function on the same basis.

In light of the expected increased use of low-carbon electricity technologies, the following unresolved issues were identified with respect to the life cycle environmental assessment of electricity generation: Inter-study variability, data availability and uncertainty, spatial and land-use issues, and intermittency of renewable resources. Masanet et al. (2013) identify as key sources of variability, among others, the assumptions on power plant efficiency and capacity factor. The capacity factor is a measure of the utilization of a power plant and therefore an important parameter in estimating the production of useful energy during its lifetime. In addition, it has been shown that variation in input data (e.g. due to regional differences between technologies, supply chains, or feedstock) can potentially alter the comparative environmental assessment (Heath and Mann 2012; Hertwich et al. 2015).

One of the largest challenges lies in balancing supply and demand under high-renewable regimes. Little is known of the additional environmental impacts of energy storage technologies, which can be used as a potential solution for balancing the grid. To understand the environmental impacts, one needs to understand in more detail the operation and deployment of individual technologies within the electricity system. This in turn influences the capacity factor. The deployment of power plants is governed by economic objectives. In a centralized power market, electricity generators with low short-run marginal cost will take precedence in system dispatch (Stoft 2002; Cludius et al. 2014). Thus, the capacity utilization of individual technologies is both dependent on the demand for electricity, the (intermittent) resource availability, as well as the technological make-up and cost-profiles of other installed capacity in the electricity system.

Each of the above described issues is addressed in one or more of the research questions (RQs). In RQ1 the effect of variation in input variables and assumptions is addressed. To provide more insight into the life cycle GHG emissions of fossil fuel electricity technologies, I focus on the effect of uncertainty and variability in the methane emissions factor of fossil fuel extraction. Different elements of issues related to intermittency issues are addressed in the following three RQs. Additional environmental impacts related to an energy storage technology are addressed in RQ2. Compressed air energy storage (CAES) was chosen as a case study for exemplifying storage impacts. The variability in capacity factor as a result of economic dispatch (satisfying demand with variable supply at lowest cost), and its influence on life cycle impacts, is addressed in RQ3. Finally, in RQ4, I address the prospective environmental impacts of the electricity system as a whole, under high-renewable regimes.

The research questions are formulated as follows:

1) What is the contribution of conventional and unconventional fossil fuel extraction processes to the climate change impacts of electricity generation? (*Paper I*, section 3.1)
2) What are the additional environmental impacts when offshore wind power is balanced with compressed air energy storage? (*Paper II*, section 3.2)
3) How does economic dispatch influence the power plant capacity factor and how does this affect the life cycle assessment results of electricity generation technologies? (*Paper III*, section 3.3)
4) What are the prospective environmental impacts related to high-renewable electricity regimes and how does this influence the setting of renewable energy targets? (*Paper IV*, section 3.4)

To answer the research questions, I build on the work of Hertwich et al. (2015). A comparative assessment that can be used for upscaling results to electricity system level has to be done in a common LCA modeling framework. The model framework (Gibon et al. 2015; Hertwich et al. 2015) contains both life cycle inventories, required for comparative analysis, and the regionalization necessary to address spatial issues in LCA modeling. For the work in *Papers III* and *IV,* an economic dispatch model was connected to the LCA model in order to capture the balancing of electricity supply and demand, while operating in a high-renewable regime.

1.4 Structure of the thesis

The remainder of this thesis is organized as follows. In the subsequent Chapter, I will present the background theory related to (hybrid) Life Cycle Assessment, which has been used in *Papers I-IV*, followed by a description of the economic dispatch modelling used for *Papers III* and *IV*. The four primary papers are summarized and discussed in Chapter 3. In Chapter 4, I will indicate my perspective on the contributions of this thesis, followed by a discussion on the limitations of this thesis. I present an outlook on the possibilities for future development of the environmental assessment of climate change mitigation options within the electricity sector. I conclude this thesis with a summary of the main conclusions of the papers and the overall main conclusion. *Papers I-IV* and *Supplementary Paper I* are included as appendices A-E.

2 Methods

'...all models are wrong, but some are useful' (Box, 1987)

Appended to this thesis are five journal articles that are intended to be independently readable works. Each of these articles contains a methodological section that should be sufficiently explanatory for an audience with experience in Industrial Ecology and its associated tools and methods such as Life Cycle Assessment. However, the form of a scientific journal article determines and limits to a certain extent the shape of its method section, and leaves little space for methodological reflections.

In this Chapter, I would like to take the opportunity offered to me by the framework of an academic thesis to discuss some of the methodological fundamentals that are at the basis of this work. In the following sections I will discuss Life Cycle Assessment (LCA), different forms of Hybrid Life Cycle Assessment (HLCA), and electricity system modelling on the basis of economic electricity dispatch models. I will indicate where each of the methods is used in appended articles.

2.1 Life Cycle Assessment

LCA is a tool that aims to analyze the environmental burden of products, services, or processes by quantifying a 'compilation and evaluation of the inputs outputs and potential environmental impacts of a product system throughout its life cycle' (Guinée et al. 2002; ISO 14040 International Standard 2006). LCA as a tool was formalized in the ISO 14000 series and generally four phases are considered, i) goal and scope definition, ii) inventory analysis, iii) impact analysis, and iv) interpretation.

In the goal and scope definition, the LCA project at hand is outlined. The most important part of this phase is to define the function of the product system under investigation and corresponding functional unit. The functional unit allows analysts to compare different product systems on a functional equivalent basis.

Inventory analysis is the most data intensive part of LCA and consists of a detailed accounting of economic activity that can be broken down to products, services and processes, in a Life Cycle Inventory (LCI). In addition to the flows between processes, resource use from the environment by the economy, as well as emissions from the economy to the environment are accounted for. These are referred to as environmental stressors. In order to facilitate the collection of data, the LCA practitioner can make use of an LCI database, which contains a large amount of process data. Several inventory databases exist, and Ecoinvent is the one most used in scientific literature (Frischknecht et al. 2005). Within the inventory, we generally consider a *foreground* and a *background* system. The foreground system consists of the model that the LCA practitioner models himself. The background system is generally the LCA database. By using the database, flows are connected from the foreground to the background system.

In the impact assessment phase of LCA, the *potential* life cycle environmental impacts are calculated. The LCI contains a list of the output per unit of final demand, for the foreground system, background system, and environmental interactions (through resource use and emissions) that is far too long to comprehend. Impact assessment seeks to reduce this list of stressors to environmental impact categories, by aggregating the contribution of stressors to pre-defined impact categories. This aggregation is done by means of a characterization factor, which characterizes the contribution of each stressor to an impact indicator representative for an impact category. It is important to stress the word potential (impact) here. Most LCAs are prospective in nature, rather than retrospective, and therefore one cannot adequately quantify the actual occurred associated environmental impact. Another factor contributing here is the associated uncertainty of the background system and the impact pathways.

The performance of LCA is inherently iterative in nature. One never goes through all phases in a single iteration. Instead, scope definition might be revisited based on the availability of more and better data, which in turn leads to a new impact assessment calculation and subsequently a new interpretation of results. All of this is part of the interpretation phase.

Several types of life cycle assessment exist. LCAs are often characterized as being either attributional (ALCA) or consequential (CLCA). In ALCA, all pollution and resource use associated with a product system is attributed to the product. No clear definition of CLCA exists, but it aims to model the consequences of changes in the life cycle as a result of different potential decisions, often focusing on marginal production technologies and their effect on demand (Hertwich 2014; Zamagni et al. 2012). As Suh and Yang (2014) point out, the distinction between ALCA and CLCA is an ideal one, and in published LCA studies we can observe a continuous spectrum between two extremes instead of a distinct dichotomy. The studies presented in this work I would classify as attributional LCA, in the sense that impacts are attributed to product systems at different levels of aggregation. In addition, based on the type of input data and method of calculation we can define process-based, EEIO based, or hybrid LCA. I will discuss the latter in the further section of this Chapter.

Mathematically, the economic activity associated with the fulfilment of the final demand (expressed in a quantity of the functional unit) can be captured by the following equation[7]:

$$\boldsymbol{x} = A\boldsymbol{x} + \boldsymbol{y} \tag{1}$$

which can be rewritten as:

$$\boldsymbol{x} = (I - A)^{-1}\boldsymbol{y} \tag{2}$$

where A indicates a (direct) technological requirements matrix, $\boldsymbol{y}$ is a column vector representing the final demand imposed on the system, and $\boldsymbol{x}$ is a column vector representing the total (economic) output. I indicates an identity matrix of appropriate size.

[7] In this thesis I employ the following conventions regarding the notation of matrix algebra. Capital variables indicate matrices, lowercase bold variables indicate (column) vectors, and lowercase variables indicate scalars. A circumflex over a vector indicates its diagonalization.

For both LCA and input-output analysis (IOA) these equations are the same. The difference lies in the construction, and scope of the A-matrix. An IO-based A-matrix can be constructed using a representation of all economic transactions between economic sectors within a certain year, indicated by *Z*, rather than bottom-up as is done in LCA.[8] It is only valid for a single economy and, as it represents current economic activity, does not include a life cycle perspective. As a result of this top-down approach, IOA prevents double counting when final demand is up-scaled to annual final demand.

Environmental stressors are recorded in a stressor matrix *S*. The correspondence between environmental stressors and impact category is represented by the characterization matrix *C*. The environmental impact ***d***, related to the total output of the product system in fulfilment of the final demand, is therefore calculated as follows.

$$\mathbf{d} = \mathrm{CS}\boldsymbol{x} = CS(I - A)^{-1}\boldsymbol{y} \qquad (3)$$

where ***d*** is a vector of which each element corresponds to a different impact category.**[9]**

2.2 Hybrid Life Cycle Assessment

As detailed as life cycle inventory databases can be, it can be argued that conventional process-based LCAs could underestimate parts of the potential environmental impacts, in particular for products where emissions occur in upstream processes (Finnveden et al. 2009). Process based LCA results were found to be underestimating impacts compared to environmentally extended IOA[9] studies, and though LCA data has more overall detail, some sectors of the economy are found to be represented with more precision in EEIO databases (Majeau-Bettez et al. 2011). Several practitioners combined IOA and process analysis leading to the emergence of hybrid LCA. Moriguchi et al. (1993) are accredited for pioneering the hybrid LCA technique (Finnveden et al. 2009). Different forms of hybrid assessment can be identified and the main ones are tiered-hybrid assessment, IO-based assessment, and integrated hybrid assessment

[8] Calculation of *A* from a full transactions matrix *Z*: $A = Z\,\hat{\boldsymbol{x}}^{-1}$

[9] Similar to LCA, IOA can be extended with a stressor matrix. We refer to this as environmentally extended input-output analysis (EE-IOA).

(Suh and Huppes 2005). A full discussion of each of the hybrid assessment forms is outside the scope of this thesis. In summary, a tiered-hybrid approach allows for complementing process-based LCAs by adding an IO based section to account for missing inputs (Stromman et al. 2009). A breakdown of the requirements matrix A in the individual sub-matrices would result in the following form:

$$A = \begin{bmatrix} A_{ff} & 0 & 0 \\ A_{pf} & A_{pp} & 0 \\ A_{nf} & 0 & A_{nn} \end{bmatrix} \tag{4}$$

Where *f*, *p*, and *n* indicate respectively the foreground, process based background, and economic background.

In an IO-based hybrid approach (Joshi 2000) the background LCA database is replaced with an IO database and the (process-based) foreground is connected to this. In matrix terms:

$$A = \begin{bmatrix} A_{ff} & 0 \\ A_{nf} & A_{nn} \end{bmatrix} \tag{5}$$

The integrated hybrid approach is based on the notion that when the foreground system is so large it potentially influences the background economy, the representation of the background economy has to be modified (Suh 2004; Peters and Hertwich 2006; Suh 2006). In other words, the parts that are zero in the above matrices are replaced with feedback flows from the foreground into the economy. Consequentially, in order to avoid double counting, the economic background has to be adjusted.

$$A = \begin{bmatrix} A_{ff} & A_{fn} \\ A_{nf} & \tilde{A}_{nn} \end{bmatrix} \tag{6}$$

The background does not have to be restricted to an IO database. Hertwich et al. (2015) combine both an LCA and IO database in an integrated hybrid framework, with feedback loops to both the process- and economic database.

Equation 2 is not computationally difficult to solve, but especially where solutions have to be computed for many iterations of increasingly large models, processing speed can become a constraint. It is possible to speed-up the calculation by using a computational equivalent of Taylor series expansion to approximate the matrix inversion. The advantages of such an approach are described by Peters (2007). In three of the *Papers* (*I, III*, and *IV*) I use this concept to speed up the computation of environmental impacts for a large number of similar, but different, inventory models. All assessments in this thesis have been carried out using either process-based (*Paper II*) or tiered-hybrid approaches (*Papers I, III,* and *IV*) and for further information regarding the methods used I refer to the Methods sections of the selected articles.

2.3 Economic dispatch of electricity

Papers III and *IV* make use of an economic electricity dispatch model. In *Paper IV* this model is integrated in the LCA model. Dispatch of electricity technologies is governed by the marginal costs of operation, and based on hourly supply and demand curves it is possible to estimate the use and production of electricity generators more accurately than by assuming an average annual production. By connecting a dispatch model to the LCA model in *Paper IV* we are able to estimate the electricity production mixes endogenous to the model, without having to rely on the mixes specified in the background database, thus presenting model users with options for designing the product system in such a way that the electricity providing service can be enjoyed at significantly lower emissions levels.

Mathematically the dispatch model is formulated as a linear program in which the objective is to minimize the costs of electricity dispatch, while adhering to basic constraints with respect to the maximum available generation capacity, transmission capacity between nodes (countries), and the energy balance over each node.

For a full mathematical description of the model I refer to the supporting information of *Paper IV*. As an example of a simplified dispatch model, I summarize here an objective function, node load balance, and generation capacity constraint. An explanation of the nomenclature is given in Table 1.

The objective of the model is to minimize the expected operational costs:

$$\min_{y} z = \sum_{g \in G} \sum_{h \in H} q_{g,h}^{gen} y_{g,h}^{gen} \tag{7}$$

A dispatch model requires that at every node (i.e. country), for every hour, there is a balance between generation, storage charge and discharge, transmission to other nodes, and the load. This is captured in the node load balance:

$$\sum_{g\in G_n} y_{g,h}^{gen} + \sum_{b\in B}\{\eta_s y_{b,h}^{stor\_dscrg} - y_{b,h}^{stor\_crg}\} + \sum_{a\in A_n^{in}} \eta_a y_{a,h}^{flow} - \sum_{a\in A_n^{out}} \eta_a y_{a,h}^{flow} = \xi_{n,h}^{load}\,, \qquad n \in N, h \in H \tag{8}$$

For every hour, and every individual generator, the generation capacity cannot be higher than the maximum installed capacity. In addition, an availability factor is included in order to capture the intermittency of renewable generators.

$$y_{g,h}^{gen} \le \xi_{g,h}^{gen} \kappa_g^{gen}\,, \qquad g \in G, h \in H \tag{9}$$

Table 1: Description of sets, indices, parameters and variables related to a simple dispatch model described in Eqs. 7, 8 and 9.

Sets and indices		
Set	**Index**	**Description**
N	n	Nodes (countries)
G	g	Generators
B	b	Storages
A	a	Arcs
H	h	Operational hour
Decision variables		
Symbol	Description	
y^{gen}	Generation	
y^{stor}	Storage flow (discharge or charge)	
y^{flow}	Arc flow	
Parameters		
Symbol	Description	
κ^{gen}	Installed generation capacity	
ξ^{load}	Load	
ξ^{gen}	Generation capacity availability	
η	Efficiency (arcs and storages)	
q^{gen}	Generator marginal cost	

3 Results

In this Chapter I will summarize and discuss the main findings of the four appended primary publications

Paper I (Bouman et al. 2015a) provides an overview of the sensitivity of LCA results related to fossil fuel power generation on methane emissions factors during coal and natural gas production by using the emission factors published by the UNFCCC.

Paper II (Bouman et al. 2015b) investigates the environmental impacts of conventional and adiabatic compressed air energy storage balancing an offshore wind power plant.

Paper III (Bouman et al. 2015c) investigates more closely how different scenarios of the development of the European power sector can influence the capacity factor of power plant technologies and how this affects the impact assessment results of the individual technologies.

Paper IV (Bouman et al. 2015d) makes use of the combination of an electricity dispatch model and an LCA model to evaluate the aggregated environmental impacts of many different configurations of the European electricity system in order to study the environmental impacts of high renewable electricity regimes.

3.1 Paper I

Multiregional environmental comparison of fossil fuel power generation – Assessment of the contribution of fugitive emissions from conventional and unconventional fossil resources.

Rationale

The environmental impacts of fossil fuel power generation are relatively well described and quantified in Life Cycle Assessment (LCA) literature. Examples of literature reviewing the LCAs of fossil power stations include: (Corsten et al. 2013; Heath et al. 2014; O'Donoughue et al. 2014; Whitaker et al. 2012). Following the popularity and increasing interest in power generation from non-conventional fossil fuel resources, most notably the shale gas development in the United States of America, several authors have pointed out that emissions occurring during the fossil fuel supply chain can have a significant climate change impact (Howarth et al. 2011; Alvarez et al. 2012; Burnham et al. 2012; Weber and Clavin 2012). However, relatively little attention has been paid to fugitive emissions within the LCA literature covering fossil fuel power plants.

As a part of the ESBLET project (Environmental Sustainability Benchmarking of Low-carbon Energy Technologies) the THEMIS model (Technology Hybridized Environmental-Economic Model with Integrated Scenarios) was developed within the Industrial Ecology group at NTNU (Hertwich et al. 2015; Gibon et al. 2015). One of the main features of THEMIS is a regionalization of the life cycle inventory data, essentially recognizing that the same type of (electricity) technologies can have different impacts in different world regions. In addition, THEMIS uses updated fugitive methane emissions factors from Burnham et al. (2012).

As can be seen from Figure 4, the *per kWh* life cycle impacts from electricity production show indeed regional variation. Considering the above, the aim of *Paper I* was to make an inventory of the ranges of fugitive methane emissions available in the literature and assess the consequences of these emissions on the life cycle greenhouse gas impacts of fossil fuel power generation.

Methods

Datasets were assembled that contained regional estimates of fugitive emissions for coal, natural gas, and shale gas emissions. The main part of the data was obtained from the United Nations Framework Convention on Climate Change (UNFCCC), to which countries report greenhouse gas emissions inventories as part of the Common Reporting Format (CRF) (UNFCCC 2012). In addition to the official emissions reports, other literature sources were consulted (Burnham et al. 2012; Howarth et al. 2011; NETL 2014 ; Weber and Clavin 2012; Bibler et al. 1998; EPA 2006; NETL 2010; Saghafi 2012; Su et al. 2011; Sørstrøm 2001).

The THEMIS model (Hertwich et al. 2015; Gibon et al. 2015) was employed to model the environmental impacts. For each of the emission factors in the dataset (totaling 227 entries) the appropriate stressors in THEMIS were adapted, and GHG emissions were evaluated over a 100-year time horizon using GWP. A life cycle inventory for eight different power plants (6 coal plants and 2 natural gas plants) was created.

Results

The LCA results from the life cycle inventories reveal that fugitive emissions can vary by orders of magnitude. This is illustrated in Figure 5. Fuel chain methane emissions constitute a substantial portion of total emissions from fossil fuel power. At the same time, we see that methane emissions vary more widely than commonly acknowledged in literature. Coal methane emissions are relevant for power plants in China and Economies in Transition, as both a result from higher fugitive emissions and the increased fuel requirements related to the use of fuel with a lower energy density.

The results in *Paper I* show such wide disparities between fugitive methane emissions that one can ask whether this is only the effect of differences in geological factors, technologies, and practices. Rather, the estimates in the UNFCCC data used as input to the model are based on engineering calculations and not measurement. This gives rise to the question to what extent these estimations are erroneous.[10]

Given the large impact of methane emissions on LCA results of fossil power generation, LCA practitioners are recommended to be aware of the issue and always perform sensitivity analysis on fugitive emissions. The results also indicate that there is a large necessity for measured data regarding fugitive emissions, not only in the USA in relation to shale gas developments, but also in other parts of the world. Fugitive emissions form a large part of life cycle GHG emissions for technologies with carbon dioxide capture and storage. The life cycle emissions from a coal power plant with CCS can be as high as emissions from a natural gas power plant without CCS. Capturing and controlling these emissions during fuel extraction is of extreme importance from a climate change perspective.

[10] A recent article shows that CO_2 emissions for China are overestimated in the UNFCCC data (Liu et al. 2015).

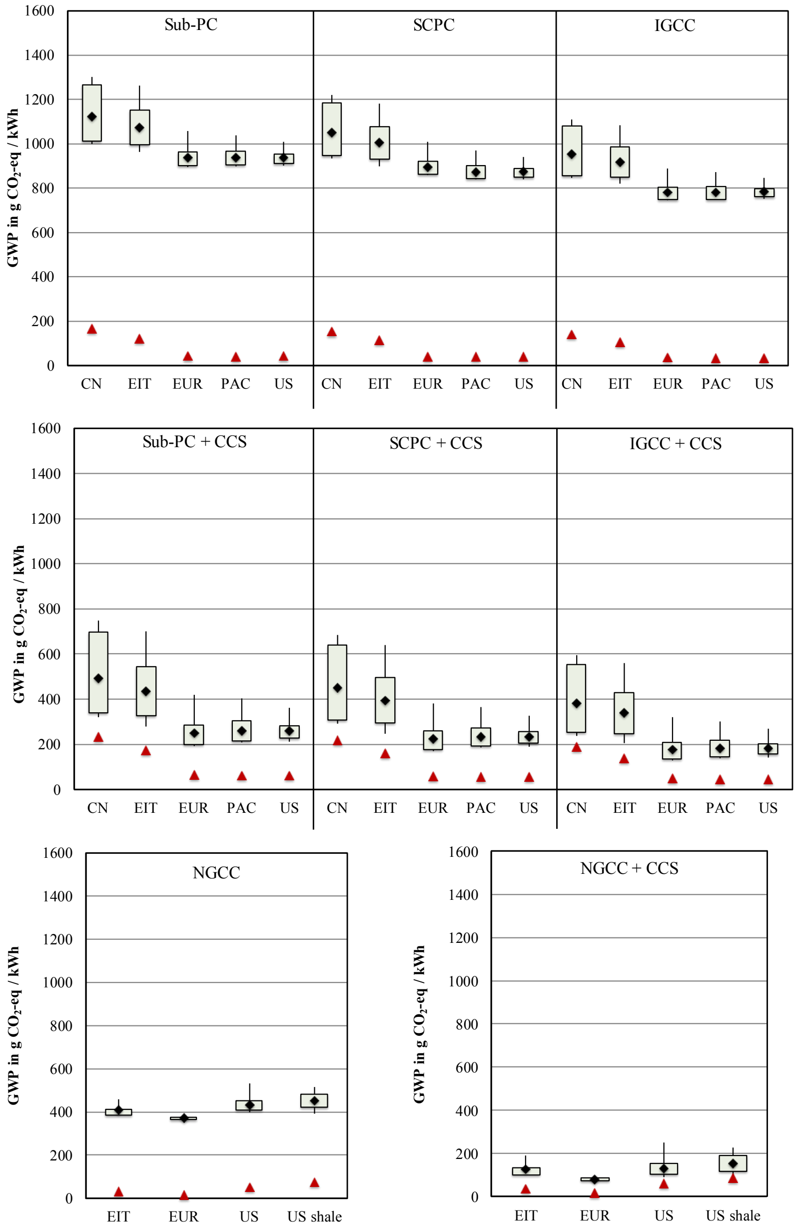

Figure 5: Calculated Global Warming Potential per kWh energy produced in sub-, supercritical, integrated gasification coal fired power plants, and natural gas fired power plants for the year 2010. Results are based on different fugitive emissions during fossil fuel extraction. Sub-PC = subcritical pulverized coal, SCPC = supercritical pulverized coal, IGCC = integrated gasification combined cycle, NGCC = natural gas combined cycle. The plotted triangles indicate the average contribution of methane emissions to the impact assessment. The plotted diamonds indicate the average GWP. Source: Bouman et al. (2015a).

3.2 Paper II

Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES).

Rationale

An often heard argument against high-renewable electricity scenarios is the intermittency related to renewable energy technologies such as wind and PV and its potential negative effects related to the large-scale application of balancing technologies. Intermittency issues can be partially solved by application of an electricity storage technology, though there are but a few technologies capable of providing utility-scale storage technology. These technologies are pumped hydro, battery storage, and compressed air energy storage (Evans et al. 2012).

In this light, *Paper II* studies the environmental impacts of compressed air energy storage with Life Cycle Assessment (LCA). Previous work on the impacts of compressed air energy storage can be found in Denholm and Kulcinski (2004) and Denholm (2005), but these environmental articles focus mainly on GHG emissions and not on other environmental impact categories. The aim of *Paper II* was to quantify the environmental impacts of various 'typical' configurations of compressed air energy storage in order to assess whether the impacts of the storage technology constituted a significant contribution to the environmental impacts of offshore produced wind power.

Methods

In conventional CAES the compressed air is heated during expansion by a natural gas turbine, which also produces electricity. It can therefore be viewed as a hybrid storage production system. In adiabatic CAES (ACAES) the thermal energy released during compression of the air is stored in a separate Thermal Energy Storage (TES), thus eliminating the need for fossil fuel combustion during the expansion phase. These two configurations of compressed air energy storage, conventional and adiabatic, were modeled in combination with three types of energy storage cavern: Aquifer storage, hard rock-mined storage, and leached salt-dome storage. Life cycle inventories were built based on the plant designs of Nakhamkin (2008) and Biasi (2009).

One of the issues related to CAES is the sizing of the system, as a larger energy storage will increase the balancing capacity of the system at the cost of greater infrastructural intensity. System scaling was done based on a fictional 'case' in which the CAES provides sufficient storage capacity to keep a 400 MW offshore wind power plant balanced. A simple model was developed that tracked the energy storage input, output, and storage level, based on a constant power demand of respectively 200 and 150 MW. The total energy storage capacity was set at two days of full wind power capacity, i.e. 19.2 GWh.

Intermittency data from Belgian offshore wind farms were used to simulate the intermittency of the offshore wind farm, for a full year and with a 15 minute resolution (ELIA 2014). The offshore wind turbine was based on the work of Arvesen et al. (2013) and Hertwich et al. (2015). The inventory was re-scaled in order to match the capacity factors related to the obtained wind power data.

Results

The results are presented in *Paper II* for both CAES and ACAES in combination with porous rock aquifer storage. In Figure 6, the foreground contribution analysis for these technologies is shown. Results for the other storage combinations are presented in the Supplementary Material of *Paper II*, which is also appended to this thesis. The construction of the storage volume contributes only a little to the overall impacts of compressed air energy storage. Instead, it is the combustion of natural gas for CAES, and to a lesser extent, the construction of the Thermal Energy Storage for ACAES that contribute most significantly to the *per kWh* impacts from compressed air storage.

The CAES is not always used, as sometimes produced wind power is sufficient to satisfy the specified target demands. In the model, between 25% and 29% of electricity from wind power is led through the storage cycle, leading to lower impacts for a balanced *wind + (A)CAES* system. The total impacts are benchmarked against impacts from the European electricity mix. Not surprisingly, the impacts of the *wind + (A)CAES* system are slightly larger than the impacts from wind power alone, the additional impacts being the price to pay for the balancing of the intermittent wind power.

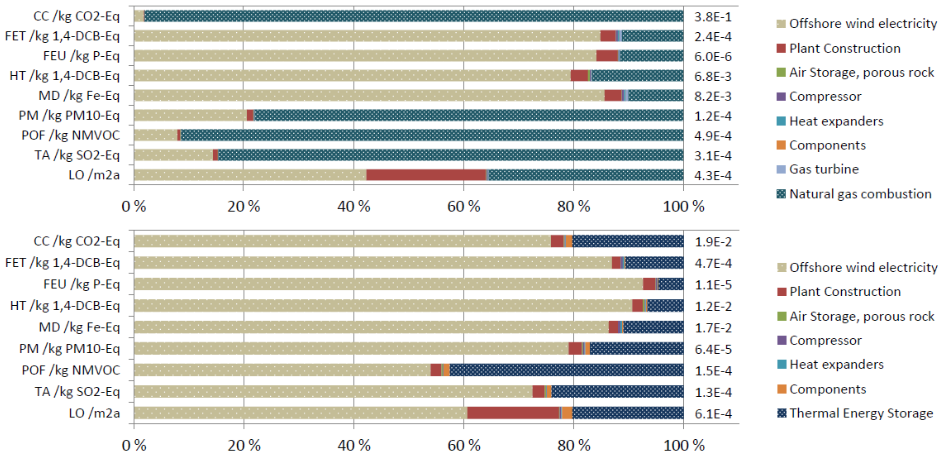

Figure 6: Contribution analysis for 1 kWh electricity generation provided by a CAES (top) and ACAES (bottom) system connected to an offshore wind power plant. Impact categories: CC - climate change, FET - freshwater ecotoxicity potential, FEU - freshwater eutrophication potential, HT - human toxicity potential, MD - metal depletion potential, PM - particulate matter formation potential, POF - photochemical oxidant formation potential, TA - terrestrial acidification potential, LO - agricultural and urban land occupation potential. Source: (Bouman et al. 2015b)

3.3 Paper III

LCA of electricity technologies using capacity factors dependent on economic dispatch.

Rationale

The calculation of Life Cycle Assessment results of electricity production requires an estimation of the electricity produced over the lifetime of the power plants. As power plants do not operate at full capacity for their entire lifetime, their 'ideal' maximum production cannot be achieved. The utilization of the maximum capacity is captured in the capacity factor. The capacity factor is ratio of a power plants actual electricity output during a period of time (e.g. 1 year), to its theoretical output at maximum capacity for the same period. For fossil fuel power plants it lies typically above 75% and for renewables mostly between 20% and 40%, as a result of the inherent variation in both solar and wind availability. Capacity factors can be used to classify power plants in terms of base load (CF=0.8-1), intermediate load (CF=0.2-0.8), and peak load (CF=0-0.2) (Raichur et al. 2015).

Paper III investigates more closely the influence of the capacity factor estimation on the environmental impacts, making use of the estimations generated by a stochastic electricity capacity expansion model (Skar et al. 2014) for various scenarios regarding the development of the European electricity sector. Modeling the electricity generation, dispatch, and investment in the power sector gives an approach to the capacity factor estimation from an economic perspective, rather than from a physical perspective.

Methods

EMPIRE is a capacity expansion planning model of the European electricity sector. Its objective is to minimize the discounted sum of investment and expected operation costs of the power sector. For each scenario and investment period, EMPIRE computes the optimal dispatch of the European electricity system (Skar et al. 2014), and as such provides an estimation of the capacity factor of individual electricity technologies. The EMPIRE model was run with different constraints in order to obtain a dataset of

36 different scenarios. For each of these scenarios, a time series of capacity factors per technology was calculated. The results of the capacity factor calculation are shown in Figure 7. Of particular interest is the gradual lower capacity of conventional coal fired power plants as a result of increasing penetration of renewables and appearance of fossil capacity with CCS technology. As power plant dispatch order is governed by the short-run marginal cost of production, these technologies will take precedence in the dispatch, which results in a lower utilization of conventional coal technology. This is known as the merit order effect (Cludius et al. 2014).

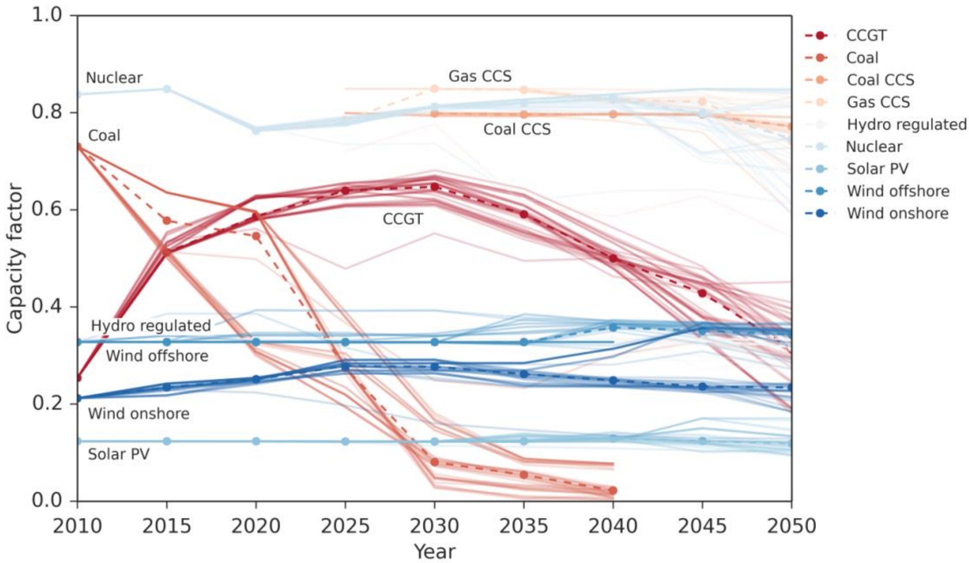

Figure 7: Capacity factors over time as calculated by the EMPIRE model for various scenarios. The median of the dataset is shown with dotted lines, and results for each individual scenario in a faded color. Abbreviations: CCGT = combined cycle gas turbine; CCS=carbon dioxide capture and storage; PV = photovoltaic. Source: Bouman et al. (2015c).

An average capacity factor was calculated by integrating the time series and dividing over the technology lifetime. Subsequently, these average capacity factors were used to re-calculate and adjust those coefficients in the foreground requirements matrix relating processes dependent on capacity factor.

Results

Across scenarios, capacity factors were found to be lower than those assumed in published LCA studies. The largest deviations were found for photovoltaic power, offshore wind power, and conventional fossil fuel power. Impact intensities were calculated for both renewable and conventional electricity generation technologies, with the average capacity factors as estimated by each EMPIRE scenario. The impacts were normalized against the default THEMIS impact results, and the resulting ratio is shown in Figure 8. Significant deviation from the 1:1 ratio implies two effects. i) The capacity utilization according to economic dispatch is different from the utilization assumed in the THEMIS model. ii) Impacts associated with power plant infrastructure dominate the life cycle. In addition, the range corresponding to the variability of capacity factors across scenarios is plotted. This can be most clearly seen in the results for onshore wind and IGCC technology without CCS. Figure 8 gives an indication of the technologies and impact indicators that are sensitive to changes in the capacity factor. For example, for the PV technologies individual impact categories cannot be identified, suggesting that all impact categories are equally sensitive to changes in the capacity factor. As the majority of the impacts of PV technologies occur during the construction phase, this is not a surprising result. A significant change can be seen in the impacts related to conventional fossil technologies. Life cycle GHG emissions are not severely influenced by changes in the capacity factor as these are mainly connected to the operational phase of the life cycle. However, terrestrial acidification, photochemical oxidant formation, and mineral depletion are impacts occurring during construction of the power plant, and appear therefore to be sensitive to changes in the capacity factor.

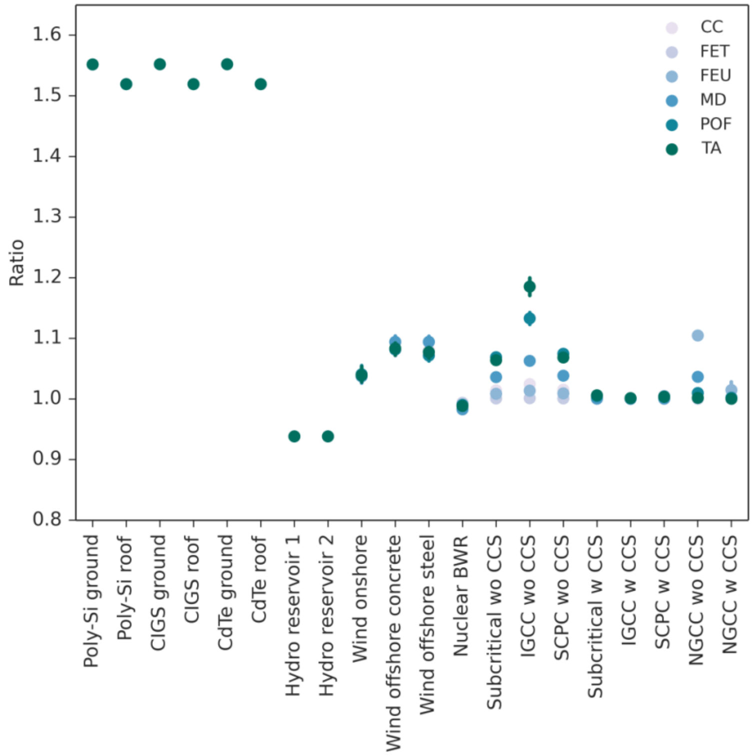

Figure 8: Ratio of life cycle impact intensities of electricity technologies calculated using CFs from EMPIRE to impact intensities calculated using THEMIS for selected impact categories. Abbreviations: CC = climate change; FET = freshwater ecotoxicity; FEU = freshwater eutrophication; MD = mineral resource depletion; POF = photochemical oxidant formation; TA = terrestrial acidification. Poly-Si = polycrystalline silicon; CIGS = copper indium gallium selenide; CdTe=cadmium telluride; Subcritical= subcritical pulverized coal; IGCC = integrated gasification combined cycle; SCPC = supercritical pulverized coal; NGCC = natural gas combined cycle; Hydro 1 / 2 = type of reservoir used as a higher/lower estimate; wind offshore concrete = offshore wind turbine with gravity-based concrete foundation; wind offshore steel = offshore wind turbine with steel-based foundation. Source: Bouman et al. (2015c)

3.4 Paper IV

Specific renewable energy technology targets can reduce life cycle impacts of electricity generation.

Rationale

In Hertwich et al. (2015) (appendix E) the THEMIS model was used to calculate the environmental impacts of low-carbon electricity generation technologies. The influence on fugitive emissions on fossil power generation was investigated in *Paper I* (appendix A, section 3.1), and the addition of compressed air energy storage to balance offshore wind power was investigated in *Paper II* (appendix B, section 3.2). However, power plants operate in a grid. In *Paper III* (appendix C, section 3.3), it is shown that the capacity factor of power plants can be of significant influence in the determination of the environmental impacts of the power plant. The capacity factor of a power plant is a direct result of the amount of operational hours during its lifetime, and an adequate estimation of the capacity factor is therefore of interest when assessing the systemic environmental impacts of electricity generation technologies.

In the previous three papers, environmental impact intensity is calculated for a *per kWh* functional unit. It has to be kept in mind that electricity produced by wind is different from electricity produced by coal fired or nuclear power. There is no difference in the physical aspects of the electricity, but its method of generation has consequences for the electricity system that are not well reflected by the employment of a functional unit on a *per kWh* basis. The intermittency of renewable generators complicates the attribution of impact results; Should the impacts of a reduced fossil capacity necessary to keep the power system stable, be attributed to wind power or remain attributed to fossil power (as was done in *Paper III*)? It is therefore of interest to investigate the life cycle impacts related to the electricity system as a whole, rather than for individual technologies.

LCA practitioners are used to present results in terms of different impact categories. However, in scoping the conclusions it often cannot be avoided to make a largely normative statement in determining the better technology in a comparative sense. Moreover, 'environmental impacts' is a term used for results from a lot of different LCA forms, ranging from a simple comparative inventory of GHG emissions, to a full LCA covering many different impact categories. Even though many practitioners tend to present a balanced discussion of impact results, often one single choice is expected in a comparative assessment. It is my belief that there can be methods to guide the LCA practitioner in this selection, or at least make the public aware of the choices practitioners have made.

The underlying theme in most of this thesis has been to look at non-climate impacts of electricity generation technologies, by harnessing the power of LCA and its capabilities of quantifying life cycle impacts of a large number of environmental stressors that can be grouped into a multitude of impact categories. A quantitative comparative assessment can only present a clear choice by somehow reducing this even further, for example through aggregation of these impact categories into a single indicator. One cannot escape a largely normative aspect in this procedure.

The aim of *Paper IV* is to bring together what is discussed above. It focuses on the following questions: When can the potential adverse environmental effects of renewable energy technologies not be ignored? How can life cycle assessment be used to inform about the systemic environmental impacts related to the electricity system? Can the setting of renewable energy targets be improved? In addition, the paper exemplifies the use of four different impact weighting strategies to stress the normative aspect in comparative assessment.

Methods

A linear dispatch model was developed for the European electricity sector. The dispatch model was based on the EMPIRE model (Skar et al. 2014), of which the results were used as input for *Paper III*. A routine to generate scenarios for installed generation capacity in Europe was used to create inputs for the dispatch model. The dispatch model provides estimates of annual electricity production and capacity factors for the individual technologies, which serve as input for the LCA model, to adjust coefficients in the foreground requirements matrix, and specify the final demand. Thus, the environmental impacts related to annual electricity production were calculated. These were subsequently aggregated using four different impact weighting methods (Goedkoop et al. 2013; Huppes et al. 2007; Soares et al. 2006; Weidema 2014). The weighting methods represent different perspectives, with foci on environment (w1), environment and resources, including land occupation (w2), human health (w3), and costs (w4).

Results

The results of the analysis are depicted in Figure 9. Each of the data points in the Figure represents an installed capacity scenario. Aggregated impact for the four different weighting methods is plotted against the fraction of renewable energy production. The trend is indicated with a scatterplot smoothing function. Increasing the fraction of renewable electricity production decreases the aggregated environmental impacts. However, over the full range of generation fractions modeled, a large variability in results was observed. Thus, there are configurations of the electricity systems that exert a lower environmental pressure than other configurations that achieve a similar renewable electricity production level. A few scenarios with low impacts are highlighted in *Paper IV* (see also Table 2 in the Paper). These capacity scenarios all resulted in a renewable electricity production share around 64%. CCS technology was present in some low-impact scenarios, but not all. A common factor among investigated scenarios was that the installed wind capacity (onshore and offshore) was significantly larger than the installed PV capacity. These observations are relevant when designing optimal low-impact electricity systems. The targets with respect to electricity generation are set in relation to the fraction of renewable energy (see e.g. the European Renewable Energy Directive (EP and CEU 2009)), but do not specify the type of technology. On the basis of the results, we suggest a greater attention for the benefits related to specific mixes of electricity technologies.

In the model, installed capacity is increased almost three-fold to 2300 GW while demand is maintained constant. Interestingly, the amount of installed capacity seems to be of little influence on aggregated impacts. This suggests that the benefits of constructing renewable infrastructure to replace generation in fossil fuel power plants greatly outweigh the costs and that significant overcapacity does not constitute an environmental issue.

By shifting perspective from individual technologies to the entire electricity system in a common framework, the potential for double counting impact intensities as a result of inconsequent attribution is also eliminated. This can occur when investigating marginal technologies and renewable technologies are credited with e.g. reduced fossil production, or discredited with the necessity of increased fossil capacity.

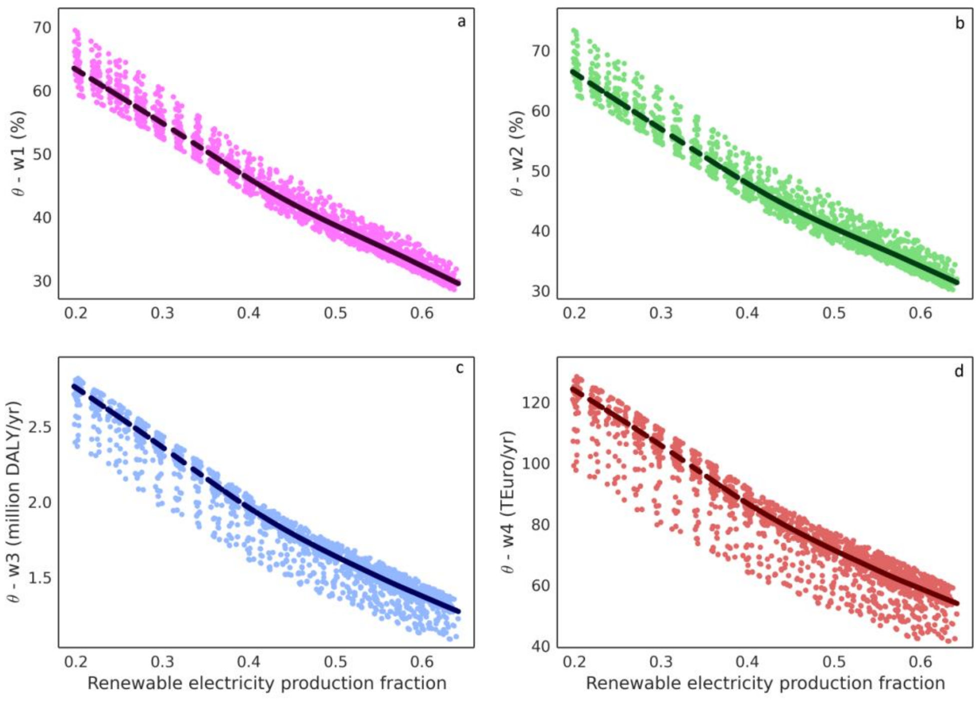

Figure 9(a-d): Aggregated impact θ as function of fraction of renewable energy production. Each of the panels indicates a different weighting method for the same set of generation capacity scenarios. The trend in each scatterplot is indicated by a lowess curve with a smoothing factor of 0.3 (Cleveland 1979). a) w1 – environmental perspective; b) w2 – environmental and resource perspective; c) w3-human health perspective; d) w4 – monetary perspective. Source: Bouman et al. (2015d).

4 Discussion

This thesis is written against the backdrop of the ongoing transition in the electricity system and its consequences for the environment. In the following section, I will articulate what are, in my opinion, the scientific contributions of the thesis. I identify contributions of a two-fold type, i.e. empirical findings in relation to the research questions investigated in the appended papers, as well as contributions from a methodological perspective. I continue with a general discussion of the limitations of the work presented in this thesis and conclude this Chapter with an outlook of ongoing developments in the field of Industrial Ecology that relate to possible extensions of this work.

4.1 Contribution

As a whole, this thesis contributes to a better understanding of the prospective environmental impacts of electricity technologies. I would like to stress that there is no such thing as *the* environmental impact of an electricity technology. Impacts can be differently categorized, and vary in magnitude based on variation and uncertainty in data and modeling. This thesis addresses this issue by highlighting the ranges of impacts from various perspectives and with various levels of detail. It demonstrates that impacts of electricity production are influenced by region, practice, and uncertainty in measurement (*Paper I*), differences in technological configuration (*Paper II*), and by the system in which the technology is embedded (*Papers III* and *IV*).

Despite the sometimes high variability in impacts, this thesis demonstrates that the impact reductions, achieved by the implementation of low-carbon electricity technologies, more than compensate for the potential adverse environmental impacts related to the implementation of these technologies. This conclusion does not change when taking the issues with generation from variable resources and economic dispatch into account. The additional environmental pressures associated with the need for balancing supply and demand appear to be relatively limited.

This does not preclude the necessity of research into the reducing the impacts of low-carbon electricity technologies. In *Paper IV*, it is shown that there are limits to the impact reduction potential achieved with aggressive development of low-carbon electricity technologies. In order to remain within a climate change scenario consistent with a 2 °C pathway, with increasing demands for electricity, impacts of electricity technologies most likely have to be reduced further. LCA can aid in properly identifying these impacts and potential areas for reduction strategies.

In *Paper I*, it is shown that the contribution of fugitive methane emissions to total life cycle GHG emission of fossil fuel power ranges from 3% to 56%, depending on region and technology. *Paper I* demonstrates that fugitive emissions vary more widely than commonly acknowledged in LCA literature, which is largely focused on USA. The large variability in the fugitive emissions data of the UNFCCC (UNFCCC 2012) indicates a necessity for more measured data in different world regions. At the same time, there is a mitigation potential associated with fugitive emissions control at coal and natural gas extraction.

In *Paper II*, we show that the environmental impacts, related to balancing the electricity production from offshore wind, are very low for almost all impact categories. It was found that the majority of impacts can be attributed to either the combustion of natural gas in conventional compressed air systems, or the thermal energy storage in adiabatic systems. The life cycle GHG emissions related to CAES were quantified previously (Denholm 2005; Denholm and Kulcinski 2004), but to our knowledge a full LCA study, covering multiple impact categories, was not published in literature. The paper thus presents a small, but useful contribution, to the body of literature concerning the environmental impacts of storage technologies.

Where LCAs of electricity technologies are usually performed on a plant-by-plant basis, the electricity system is explicitly taken into account in *Paper III*. Using the capacity utilization estimations of the EMPIRE model (Skar et al. 2014) it is shown that the capacity factor of power plants can have a large influence on the comparative assessment of impact intensities of different electricity technologies. In addition, it is demonstrated that the sensitivity for changes to the capacity factor is different for both technologies and impact categories, depending on the stage in the life cycle (e.g. construction, operation) where impacts occur. *Paper III* presents a contribution in showcasing that a better understanding of the operation of the electricity market can help in determining a more accurate estimation of environmental impacts.

While the comparative assessment of electricity technologies can be influenced by differences in capacity factor estimation, it is shown in *Paper IV* that the environmental advantages of low-carbon technologies over fossil technologies remain large. The results in *Paper IV* suggest that the overcapacity that is present in high-renewable regimes reduces the marginal benefit of additional capacity, but still contributes to such a benefit for the entire range of renewable capacities investigated. Nonetheless, *Paper IV* presents a valuable contribution by explicitly showing that even between renewable scenarios that are similar on an aggregate level, there are large variations possible in resulting impact. For the investigated low-impact scenarios, it was found that installed wind capacity was significantly larger than installed PV capacity. This could inform the setting of renewable energy targets motivated by impact reduction. In addition, the paper draws attention to the normative aspect of comparative LCA, which is often overlooked, by showing results for four different impact weighting methods.

From a methodological perspective, this thesis addresses some of the issues that were identified with respect to the environmental assessment of electricity technologies in Chapter 1: inter-study variability, spatial issues, and impacts due to intermittency of variable resources. The life cycle models used in this thesis were all custom-built and written in either Matlab or Python programming language. Contrary to the use of dedicated LCA software, a custom model that is based on the Leontief framework gives the user far more flexibility in manipulating input data, calculations, and the output of results. For example, the assembly of fugitive methane emissions data for *Paper I* resulted in a total of 227 data entries. Manually constructing a separate LCI model in LCA software, such as SimaPro (Pré 2015) or CMLCA (Heijungs 2012), for each of these entries would have been a painstaking and time consuming process. Instead, the adjustment of the environmental stressor matrix, by coupling the assembled dataset to the inventory, using the regionalization provided by the THEMIS model (Gibon et al. 2015; Hertwich et al. 2015), and subsequent calculation of life cycle impacts, allowed for the possibility of generating ranges of results for different regions. Other authors have produced ranges of impact results related to fugitive emissions, but these are connected to probability distributions, e.g. in combination with Monte Carlo analysis, rather than to an exogenous dataset (Burnham et al. 2012; Weber and Clavin 2012).

The concept of adjusting matrices on the basis of external data was again used in *Paper III. T*he capacity factors generated by the EMPIRE model (Skar et al. 2014) were used to adjust coefficients in the technological requirements matrix, thus allowing to re-use the 'typical' inventories, but adjust them with varying key parameters. Both *Paper III* and *IV* contribute to the showcasing of combining an LCA model with economic dispatch modelling, and it is shown that dispatch modelling provides opportunities to assist in the estimation of key parameters in LCA of electricity technologies.

Paper IV presents an opportunity to quantify both direct and indirect impacts associated with *any* configuration of the European electricity system, at a level that has not been shown previously in literature. Dispatch models, partial equilibrium models, and integrated assessment models concerning the energy sector, have been dealing with environmental impacts mostly through the application of emission factors, which are independent of model output. Often, these emissions factors only represent direct impacts for a limited amount of impact categories.

Paper IV takes a similar approach as is described in Pehnt et al. (2008), but differs on some distinct points. Pehnt et al. (2008) link a stochastic model of the European electricity market (E2M2s) to an LCA model to study the impacts of additional wind power production in Germany. It is unclear to what extent the electricity production estimates by E2M2s are reflected in the calculation of infrastructural impact intensity, as is done in *Paper IV*. In addition, the authors take a substitution approach and subtract replaced fossil fuel power impacts from the calculated wind power impacts. This leads to the quantification of an impact reduction potential, rather than a quantification of the impacts associated with operation of the electricity system as a whole. As such, it implies causality between the introduction of wind and displacement of fossil power that is perhaps misleading as pointed out by Hertwich (2014).

4.2 Limitations

The construction of life cycle inventories and input-output databases is both time and data-intensive work. Given the available time and scope of this project, a balance had to be found between the detail of the data inventories, the construction of the underlying models, and the exploration of the research questions. In this section, I will summarize the limitations regarding the case studies and methods applied in this work.

My first comment would be with respect to the technological coverage of the work. The original set of (foreground) life cycle inventories for the article by Hertwich et al. (2015) consists of 21 technologies. Each of these inventories was based on technology descriptions of state-of-the-art 'typical' power plants, in line with the aim of the article to estimate potential impacts of technologies likely to be adopted at large-scale in the coming years. As a result, up-scaled impact calculations of the modeled technologies could underestimate the environmental impacts of the current fleet, which are arguably higher due the inclusion of older technologies. Conversely, some of the data available in the background ecoinvent database is over 20 years old, which implies that the impacts related to background processes are not up-to-date. Solving this issue was the main motivation for the integration of the THEMIS model to calculate impacts for 2030 and 2050 (Hertwich et al. 2015). In the later stages of this project the set of foreground inventories was expanded with inventories for a nuclear boiling water reactor, based on Dones et al. (2007), an adaptation of a CHP plant for biomass, based on Singh et al. (2014), and inventories of compressed air storage (*Paper II*) and Li-ion batteries (Ellingsen et al. 2014).

From an LCA-perspective, the (growing) set of foreground inventories available in connection with the THEMIS model is noteworthy. However, the scenario outputs used as input for the LCA model in *Paper III* and *Paper IV* contain a technological coverage that is simultaneously less detailed and more comprehensive, i.e. coal-fired power is modeled as a single technology, but oil- and lignite-fired power plants (for which there are no foreground inventories available in THEMIS) are also included. This disparity necessitates the use of concordance matrices in order to connect the scenarios with the LCA modeling. In lack of better information, an equal share between technologies of the same type (e.g. coal without CCS, PV) was assumed, but especially where impacts between technologies of the same type differ significantly, a better estimation of production volumes of all technologies within the (foreground) life cycle inventory is warranted.

The adaptation of a linear dispatch model in *Paper IV* has advantages with respect to computational speed and resolution at which the model can operate. At the same time, costs are determined exogenous to the model, irrespective of system configuration and operation, which is an attribute inherent to using a linear model. In the interest of time and scope, sensitivity analysis was not performed with respect to the input data for the model, leading to a limitation in the robustness of results under different cost assumptions. Moreover, as Prado-Lopez et al. (2015) point out, the external normalization applied in the weighting procedures evaluates the magnitude of impacts relative to current total impact levels. As results for some impact categories might appear to be small due to the magnitude of their normalization reference, it is less suitable for the identification of significant differences between technologies when evaluating trade-offs in a comparative way.

In summary, the models used in this thesis are constrained by their technological resolution, and limited in the extent to which the research questions were explored using sensitivity analysis. They are unfit to assess the large-scale deployment of a potentially new and disruptive technology. From a climate perspective there are promising results reported with respect to bio-energy as baseload power source, especially in combination with CCS technology (Sanchez et al. 2015). An inventory for combined heat and power generation with bio-energy was included in *Paper IV*. However, a comprehensive analysis of increased use of bio-energy would require a detailed assessment of its competition with the agricultural food system, the changes in albedo as a result of harvesting, and how to consequentially evaluate the global warming potential of carbon dioxide emissions from biogenic origin (Guest et al.

2013; Cherubini et al. 2013). All of these were beyond the scope of this thesis, but I express here the hope that we will eventually be able to include a more exhaustive perspective on bio-energy within the modelling framework.

In addition to the above described limitations, there are some limitations inherently related to the use of an LCA framework. The impact assessment methods such as ReCiPe (Goedkoop et al. 2013) incorporate modeling choices with respect to e.g. impact pathways, the time horizon toxic of releases, and the assumed shape of dose-response curves. For example, quantifying a spatially defined environmental impact such as the build-up of downstream pollutant concentrations from a riverside power plant is not possible in LCA, since it is a static method in which time is not included (Yellishetty et al. 2009). In addition, where this thesis contains regionalization with respect to characteristics concerning the life cycle inventory, also the environmental impacts can vary throughout space (Mutel et al. 2012). Inventory analysis on a large regional level as conducted here cannot yet be reliably matched with regionally dependent characterization factors.

4.2.1 A note on uncertainty and variability

In LCA, uncertainty in inventory data is usually addressed by making use of a pedigree matrix. The pedigree matrix gives a semi-quantitative indication of five parameters: reliability of the data, completeness, and temporal, geographical, and technological correlation. Based on the indicator score additional parametric uncertainty (Kennedy and O'Hagan 2001) is estimated, related to the data not being of optimal quality (Weidema and Wesnæs 1996). The ecoinvent data contains pedigree matrix coefficients that provide a standard deviation to the inventory assumes an underlying probability distribution, and can be used as an input to Monte Carlo simulations.

Much of the work performed in this thesis is built upon calculating ranges of environmental impacts of electricity generation by implementing a large number of discrete variations with respect to certain elements in either the technological requirements or environmental stressor matrix. In *Paper I* the fugitive emissions are varied, in *Paper III* we apply a variety of capacity factors, and in *Paper IV* we simulate many different dispatch scenarios. These variations do not reflect parametric uncertainty one can study with Monte Carlo simulations, as a result from, for example, measurement error. Rather, much of the input data and key assumptions are based on the output of other modelling exercises. Hence, the ranges presented in this thesis reflect the variability in those exercises. By producing ranges, rather than single impact results, we illustrate a bandwidth of LCA results and its influence on the comparative environmental assessment of electricity generation technologies. One can think of this as an attempt to visualize the parametric variability with respect to LCA results, rather than the parametric uncertainty. Uncertainty is not included so as to prevent obfuscating the effects of variability. The lack of a rigorous and consequent treatment of uncertainties in the data does pose a limitation on the applicability of the models as decision making instrument.

4.3 Outlook

As has been pointed out earlier in this thesis, LCA is used to attribute environmental and resource impacts to product systems. In the previous Chapters, I have shown how the method can be used to quantify both the direct and indirect impacts related to the implementation of low-carbon electricity technologies. While the focus of policy makers seems to be mainly aimed at reducing greenhouse gas emissions, both LCA and IOA hand us the tools to quantify other potential environmental consequences of such a transition. This work fits in a trend to use life cycle models to inform policy with respect to the effects and impacts of large-scale changes within the technosphere.

However, the models are not (yet) at the level that they can pinpoint concrete detailed climate change mitigation interventions. By nature, LCA is a comparative methodology and variability and uncertainty related to input data complicates the comparative analysis of policy options. As indicated in the previous section, I would argue that systematic inclusion of uncertainty is necessary in order to assist in the comparative analysis between technologies.

At the same time, the technological coverage of the life cycle inventories of electricity technologies will continue to be expanded. A natural extension of the work presented in *Paper IV* would be to adjust the electricity dispatch model to include the capacity investment decision, thus providing the capability to generate electricity system development scenarios such as provided by EMPIRE (Skar et al. 2014). By allowing environmental impacts to influence the capacity investment decision, we would have an opportunity to show how internalization of external costs affects the electricity system development, something which is not included in current economic models. Such an approach would require making use of an integrated model as future technologies are built using a gradually cleaner electricity mix, and would allow investigating implementation strategies regarding the ideal moment to increase renewable energy deployments (also known as a front loading and back loading strategy (Ravikumar et al. 2014)).

It is my hope that the future modeling can benefit from other methodological advances in Industrial Ecology, through the use of a collaborative open source modeling framework as described by Pauliuk et al. (2015). At the time of the writing of this thesis, the model presented in *Paper IV* is not yet available as open source, but the ambition exists to release a more generally applicable form of the model to the research community.

Based on the experience with working with the background database, I would argue that the full potential of life cycle inventory data is not used. For example, as Singh et al. (2015) point out, LCA can provide information on life cycle material use, which can be easier to comprehend for a general audience the characterized life cycle impact indicators. Ideally, a life cycle assessment accounting framework fulfils not only material balances, but also energy balances, and economic (price) balances. The detail and accurateness of the underlying background dataset is of crucial importance here. Traditionally, LCA process data show a large dispersion in accuracy and detail. Some processes are quantified in minute detail, whereas others are based solely on proxy data. For the purpose of impact assessment this disparity often does not matter, but as LCA is used more and more to answer questions related to entire technological systems on a prospective basis, rather than for a simple product comparison, sufficient detail in the background is paramount. A comprehensive, hierarchically organized, life cycle inventory database, which not only contains economic flows and environmental stressors, but also accounts for energy balances and price of products in a regionalized context, would be a step forward.

In this context, it is important to switch from a technological requirements matrix perspective (where each process has only one outflow), to a perspective that allows a more flexible treatment of multi-output processes, i.e. processes that produce more than one (economic) product. For example, in *Paper I* are the greenhouse gas emissions produced by fossil fuel power with CCS attributed to power generation, even though gypsum production is a by-product of the CCS process. Though this is a defendable assumption, it would be preferable to be able to avoid the necessity of making this assumption during the construction of the life cycle inventory. Switching to a supply-use framework, in which the supply and use of products and processes are separately recorded and a technological requirements matrix is calculated by applying a construct, would provide an opportunity to resolve this issue. Interesting work in this context has been performed by Majeau-Bettez et al. (2015) who have proven the functional equivalence of constructs in IOA and allocation in LCA and describe modeling in a multilayered supply-use framework, where material, financial, and energy data are stored (Majeau-Bettez et al. 2014; Majeau-Bettez et al. 2015).

In general, I see an opportunity for the LCA community to make use of linking socio-economic models with different levels of complexity, the most integrative of them being integrated assessment models (IAMs). The strength of these models lies in their capability to provide a global coverage and integration of the energy system with climate system and the economy, but they are often criticized for their aggregation levels and coarseness (Bauer 2015). I believe that LCA has come to a level of maturity that it can contribute significantly to the comparative assessment of (energy) transformation pathways studied by these models, by quantifying consistently a range of environmental impact indicators at a level of detail previously not considered in these models. LCA is therefore indispensable as a tool for studying the effects of climate change mitigation policies.

5 Conclusions

In this thesis I have presented four papers. These papers are summarized in Chapter 3. Their contribution, the limitations, and a general outlook of the field are presented in Chapter 4. In the following paragraphs I will iterate the main conclusions of the papers as well as my final conclusions regarding this work.

Paper I

Fugitive methane emissions contribute significantly towards the life cycle GHG emissions of fossil fuel power generation. This effect becomes larger when carbon dioxide capture and storage technology is employed at the power plant, as emissions from the power plant are captured, but upstream fugitive emissions per unit generation increase as a result of the decrease in power plant efficiency. Though historically there has been a lot of attention for the magnitude of fugitive emissions in the US, there is an increased need for measurements in other countries.

Paper II

The life cycle environmental impacts of compressed air energy storage (CAES) as a means of balancing the impacts of offshore wind power were investigated. Including the environmental impacts of balancing slightly increases the impacts of wind power. When compared to the average electricity mix, however, the impacts from CAES-balanced offshore wind power are still considerably lower. Adiabatic CAES, in which heat released during compression is stored in a separate thermal energy system, was shown to have a better environmental performance than conventional CAES. This conclusion is dependent on the total storage capacity of the ACAES system, as the majority of related impacts can be attributed to the construction of the thermal energy storage.

Paper III

The influence of the capacity factor estimation on LCA results of power generation was investigated. An economic capacity expansion planning model, which included economic dispatch, was used to estimate capacity utilization for various scenarios describing the development of European power systems. It was shown that the capacity factor estimation can have a large influence on the comparative assessment of the impact intensities of different renewable energy technologies. Though the capacity factor for conventional fossil fuel technologies can be expected to fluctuate most, depending on renewable electricity penetration, the influence of the capacity factors is limited to those impacts occurring during construction of the power plant infrastructure.

Paper IV

The aggregated life cycle environmental impacts of European electricity generation were calculated for different configurations of the electricity system. It was ensured that the system balanced supply and demand through the use of an electricity dispatch model, taking constraining factors such as transmission and storage capacity into account. Individual impact categories were aggregated according to four distinct weighting methods. As a general trend, aggregated impacts decrease as function of increasing renewable power penetration. At the same time, for renewable energy penetration scenarios where the total share of renewables is equal, but individual shares of renewable technologies differ, a large spread in impact results was observed. This suggests that the renewable energy targets currently specified in relation to the power sector can benefit from specifying the mix of technologies, rather than an aggregate target.

Can we construct a low-impact electricity system?

My short answer would be: yes. From an environmental perspective it has been shown in this thesis that a high penetration of renewable electricity technologies, some of which have environmental pressures that exceed the ones exerted by conventional fossil technology, still results in a system that exhibits lower impacts than the one we have today. However, other potential bottlenecks that are not studied in this thesis might have to be overcome. For example, bottlenecks could be related to the necessary rate of expansion of transmission and production capacity, and location and capacity considerations related to wind power and large-scale energy storage. The papers in this thesis have investigated impact reduction opportunities for fossil fuel technologies, the inclusion of a storage technology to balance intermittency, have looked at the balancing of supply and demand, and the resulting overcapacity necessary for a balanced high-renewable system. Potential constraining factors such as financing, location permits, stakeholder opposition, and production capacity and time required for building necessary low-carbon infrastructure, were not investigated.

The work in this thesis does not give a definite answer as to how a low-impact system might look like in detail. The models employed are necessarily constrained in technological resolution, scope, and complexity. Novel technologies are not sufficiently represented. Both the variability and uncertainty associated with input data needs to be addressed systematically in order to make a functional comparative assessment model capable of capturing the economic, social, and environmental dynamics that would allow us to pinpoint in detail, where, when, and how to build and operate renewable power stations.

References

Alvarez, R. A., S. W. Pacala, J. J. Winebrake, W. L. Chameides, and S. P. Hamburg. 2012. Greater focus needed on methane leakage from natural gas infrastructure. *Proceedings of the National Academy of Sciences of the United States of America* 109(17): 6435-6440.

Arrhenius, S. 1896. On the influence of carbonic acid in the air upon the temperature of the ground. *Philosophical Magazine Series 5* 41(251): 237-276.

Arvesen, A., C. Birkeland, and E. G. Hertwich. 2013. The Importance of Ships and Spare Parts in LCAs of Offshore Wind Power. *Environmental Science & Technology* 47(6): 2948-2956.

Bauer, N. 2015. Power Systems: Carbon negative at the regional level. *Nature Climate Change* 5(3): 196-197.

Biasi, V. d. 2009. Fundamental Analyses to Optimize Adiabatic CAES Plant Efficiencies. *Gas Turbine World*: 26-28.

Bibler, C. J., J. S. Marshall, and R. C. Pilcher. 1998. Status of worldwide coal mine methane emissions and use. *International Journal of Coal Geology* 35(1-4): 283-310.

Bouman, E. A., A. Ramirez, and E. G. Hertwich. 2015a. Multiregional environmental comparison of fossil fuel power generation—Assessment of the contribution of fugitive emissions from conventional and unconventional fossil resources. *International Journal of Greenhouse Gas Control* 33(0): 1-9.

Bouman, E. A., M. M. Øberg, and E. G. Hertwich. 2015b. Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES). *under review*.

Bouman, E. A., C. Skar, and E. G. Hertwich. 2015c. LCA of electricity technologies using capacity factors dependent on economic dispatch. *submitted to Environmental Research Letters*.

Bouman, E. A., C. Skar, and E. G. Hertwich. 2015d. Specific renewable energy technology targets can reduce life cycle impacts of electricity generation. *submitted to Environmental Science & Technology*.

Bruckner, T., I. A. Bashmakov, Y. Mulugetta, H. Chum, A. d. l. V. Navarro, J. Edmonds, A. Faaij, B. Fungtammasan, A. Garg, E. Hertwich, D. Honnery, D. Infield, M. Kainuma, S. Khennas, S. Kim, H. B. Nimir, K. Riahi, N. Strachan, R. Wiser, X. Zhang, and 2014. Energy Systems. In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by O. Edenhofer, et al. Cambridge, United Kingdom and New York, NY, USA.

Buonocore, J. J., P. Luckow, G. Norris, J. D. Spengler, B. Biewald, J. Fisher, and J. I. Levy. 2015. Health and climate benefits of different energy-efficiency and renewable energy choices. *Nature Clim. Change* advance online publication, 10.1038/nclimate2771.

Burkhardt, J. J., G. Heath, and E. Cohen. 2012. Life Cycle Greenhouse Gas Emissions of Trough and Tower Concentrating Solar Power Electricity Generation. *Journal of Industrial Ecology* 16: S93-S109.

Burnham, A., J. Han, C. E. Clark, M. Wang, J. B. Dunn, and I. Palou-Rivera. 2012. Life-Cycle Greenhouse Gas Emissions of Shale Gas, Natural Gas, Coal, and Petroleum. *Environmental Science & Technology* 46(2): 619-627.

Cherubini, F., R. M. Bright, and A. H. Stromman. 2013. Global climate impacts of forest bioenergy: what, when and how to measure? *Environmental Research Letters* 8(1): 014049.

Ciais, P., C. Sabine, G. Bala, L. Bopp, V. Brovkin, J. Canadell, A. Chhabra, F. DeFries, J. Galloway, M. Heimann, C. Jones, C. Le Quere, R. B. Myneni, S. Piao, and P. Thornton. 2013. Carbon and Other Biogeochemical Cycles. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Reprot of the Intergovernmental Panel on Climate Change*, edited by T. F. Stocker, et al. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Clarke, L., K. Jiang, K. Akimoto, M. Babiker, G. Blanford, K. Fisher-Vanden, J.-C. Hourcade, V. Krey, E. Kriegler, A. Löschel, D. McCollum, S. Paltsev, S. Rose, P. R. Shukla, M. Tavoni, B. C. C. van der Zwaan, and D. P. van Vuuren. 2014. Assessing Transformation Pathways. In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by O. Edenhofer, et al. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Cleveland, W. S. 1979. Robust Locally Weighted Regression and Smoothing Scatterplots. *Journal of the American Statistical Association* 74(368): 829-836.

Cludius, J., H. Hermann, F. C. Matthes, and V. Graichen. 2014. The merit order effect of wind and photovoltaic electricity generation in Germany 2008-2016: Estimation and distributional implications. *Energy Economics* 44: 302-313.

Corsten, M., A. Ramirez, L. Shen, J. Koornneef, and A. Faaij. 2013. Environmental impact assessment of CCS chains - Lessons learned and limitations from LCA literature. *International Journal of Greenhouse Gas Control* 13: 59-71.

Davis, S. J., K. Caldeira, and H. D. Matthews. 2010. Future CO2 Emissions and Climate Change from Existing Energy Infrastructure. *Science* 329(5997): 1330-1333.

Denholm, P. and G. L. Kulcinski. 2004. Life cycle energy requirements and greenhouse gas emissions from large scale energy storage systems. *Energy Conversion and Management* 45(13-14): 2153-2172.

Denholm, P. K., Gerald L; Holloway, Tracey. 2005. Emissions and Energy Efficiency Assessment of Baseload WInd Energy Systems. *Environmental Science & Technology* 39: 1903-1911.

Dolan, S. L. and G. A. Heath. 2012. Life Cycle Greenhouse Gas Emissions of Utility-Scale Wind Power. *Journal of Industrial Ecology* 16: S136-S154.

Dones, R., C. Bauer, R. Bolliger, B. Burger, M. Faist Emmenegger, R. Frischknecht, T. Heck, N. Jungbluth, and A. Röder. 2007. *Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries*. Dübendorf, CH: Swiss Centre for Life Cycle Inventories.

EC, European Commission. 2015. Environmental Assessment. http://ec.europa.eu/environment/eia/home.htm. Accessed 17 Sep 2015.

Edenhofer, O., R. Pichs-Madruga, Y. Sokona, S. Kadner, J. C. Minx, S. Brunner, S. Agrawala, G. Baiocchi, I. A. Bashmakov, G. Blanco, J. Broome, T. Bruckner, M. Bustamante, L. Clarke, M. Conte Grand, F. Creutzig, X. Cruz-Núñez, S. Dhakal,

N. K. Dubash, P. Eickemeier, E. Farahani, M. Fischedick, M. Fleurbaey, R. Gerlagh, L. Gómez-Echeverri, S. Gupta, J. Harnisch, K. Jiang, F. Jotzo, S. Kartha, S. Klasen, C. Kolstad, V. Krey, H. Kunreuther, O. Lucon, O. Masera, Y. Mulugetta, R. B. Norgaard, A. Patt, N. H. Ravindranath, K. Riahi, J. Roy, A. Sagar, R. Schaeffer, S. Schlömer, K. C. Seto, K. Seyboth, R. Sims, P. Smith, E. Somanathan, R. Stavins, C. von Stechow, T. Sterner, T. Sugiyama, S. Suh, D. Ürge-Vorsatz, K. Urama, A. Venables, D. G. Victor, E.Weber, D. Zhou, J. Zou, and T. Zwickel. 2014. Technical Summary. In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by O. Edenhofer, et al. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

ELIA. 2014. Historical wind power generation data. http://www.elia.be/en/grid-data/power-generation/wind-power. Accessed 17 Sep 2015.

Ellingsen, L. A. W., G. Majeau-Bettez, B. Singh, A. K. Srivastava, L. O. Valoen, and A. H. Stromman. 2014. Life Cycle Assessment of a Lithium-Ion Battery Vehicle Pack. *Journal of Industrial Ecology* 18(1): 113-124.

EP and CEU. 2009. Directive 2009/28/EC of the European Parliament and of the Council *Offical Journal of the European Union*: L140/116.

EPA, Environmental Protection Agency. 2006. *Global Mitigation of Non-CO2 greenhouse gases*. Washington DC:

Evans, A., V. Strezov, and T. J. Evans. 2012. Assessment of utility energy storage options for increased renewable energy penetration. *Renewable and Sustainable Energy Reviews* 16(6): 4141-4147.

Finnveden, G. and A. Moberg. 2005. Environmental systems analysis tools - an overview. *Journal of Cleaner Production* 13(12): 1165-1173.

Finnveden, G., M. Z. Hauschild, T. Ekvall, J. Guinee, R. Heijungs, S. Hellweg, A. Koehler, D. Pennington, and S. Suh. 2009. Recent developments in Life Cycle Assessment. *Journal of Environmental Management* 91(1): 1-21.

Fleurbaey, M., S. Kartha, S. Bolwig, Y. L. Chee, Y. Chen, E. Corbera, F. Lecocq, W. Lutz, M. S. Muylaert, R. B. Norgaard, C. Okereke, and A. D. Sagar. 2014. Sustainable Development and Equity. In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by O. Edenhofer, et al. Cambridge, United Kingdom, and New York, NY, USA: Cambridge University Press.

Freidberg, S. 2015. From behind the curtain: talking about values in LCA. *The International Journal of Life Cycle Assessment* 10.1007/s11367-015-0879-6: 1-5.

Frischknecht, R., N. Jungbluth, H. J. Althaus, G. Doka, R. Dones, T. Heck, S. Hellweg, R. Hischier, T. Nemecek, G. Rebitzer, and M. Spielmann. 2005. The ecoinvent database: Overview and methodological framework. *International Journal of Life Cycle Assessment* 10(1): 3-9.

Frosch, R. A. and N. E. Gallopoulos. 1989. Strategies for Manufacturing. *Scientific American* 261(3): 144-152.

Fthenakis, V. and H. C. Kim. 2009. Land use and electricity generation: A life-cycle analysis. *Renewable and Sustainable Energy Reviews* 13(6-7): 1465-1474.

Garner, A. and G. A. Keoleian. 1995. *Industrial Ecology: An introduction*. Ann Arbor, MI: University of Michigan National Pollution Prevention Center for Higher Eductation

Gibon, T., R. Wood, A. Arvesen, J. D. Bergesen, S. Suh, and E. G. Hertwich. 2015. A methodology for integrated, multiregional life cycle assessment scenarios under large-scale technological change. *Environmental Science & Technology* 10.1021/acs.est.5b01558.

Goedkoop, M., R. Heijungs, M. A. J. Huijbregts, A. de Schryver, J. Struijs, and R. Van Zelm. 2013. *ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level*.

Graedel, T. E. and B. R. Allenby. 1995. *Industrial Ecology*. Upper Saddle River: Prentice Hall.

Guest, G., F. Cherubini, and A. H. Stromman. 2013. Global Warming Potential of Carbon Dioxide Emissions from Biomass Stored in the Anthroposphere and Used for Bioenergy at End of Life. *Journal of Industrial Ecology* 17(1): 20-30.

Guinée, J. B., M. Gorrée, R. Heijungs, G. Huppes, R. Kleijn, A. De Koning, L. Van Oers, A. W. Sleeswijk, S. Suh, H. A. Udo de Haes, H. de Bruijn, R. van Duin, and M. A. J. Huijbregts. 2002. *Handbook on Life Cycle Assessment. Operational guide to the ISO standards*. Edited by J. Guinée, *Eco-efficiency in industry and science*. Dordrecht: Kluwer Academic Publishers.

Heath, G. A. and M. K. Mann. 2012. Background and Reflections on the Life Cycle Assessment Harmonization Project. *Journal of Industrial Ecology* 16(SUPPL.1): S8-S11.

Heath, G. A., P. O'Donoughue, D. J. Arent, and M. Bazilian. 2014. Harmonization of initial estimates of shale gas life cycle greenhouse gas emissions for electric power generation. *Proceedings of the National Academy of Sciences of the United States of America* 111(31): E3167-E3176.

Heijungs, R. 2012. CMLCA, Scientific software for LCA, IOA, EIOA and more. www.cmlca.eu. Accessed 24 Sep 2015.

Hertwich, E. 2014. Understanding the Climate Mitigation Benefits of Product Systems: Comment on "Using Attributional Life Cycle Assessment to Estimate Climate-Change Mitigation ...". *Journal of Industrial Ecology* 18(3): 464-465.

Hertwich, E. G., T. Gibon, E. A. Bouman, A. Arvesen, S. Suh, G. A. Heath, J. D. Bergesen, A. Ramirez, M. I. Vega, and L. Shi. 2015. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. *Proceedings of the National Academy of Sciences* 112(20): 6277-6282.

Hond, F. d. 2000. Industrial ecology: a review. *Regional Environmental Change* 1(2): 60-69.

Howarth, R. W., R. Santoro, and A. Ingraffea. 2011. Methane and the greenhouse-gas footprint of natural gas from shale formations. *Climatic Change* 106(4): 679-690.

Hsu, D. D., P. O'Donoughue, V. Fthenakis, G. A. Heath, H. C. Kim, P. Sawyer, J. K. Choi, and D. E. Turney. 2012. Life Cycle Greenhouse Gas Emissions of Crystalline Silicon Photovoltaic Electricity Generation. *Journal of Industrial Ecology* 16: S122-S135.

Huppes, G., M. D. Davidson, J. Kuyper, L. van Oers, H. A. U. de Haes, and G. Warringa. 2007. Eco-efficient environmental policy in oil and gas production in The Netherlands. *Ecological Economics* 61(1): 43-51.

IEA. 2010. *Energy technology perspectives 2010: Scenarios and strategies to 2050*. Paris: OECD/IEA.

IEA. 2015. *Energy Technology Perspectives 2015: Mobilising innovation to accelerate climate action*. Paris: OECD/IEA.

IPCC. 2013. Summary for Policymakers. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, edited by T. F. Stocker, et al. Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

IPCC. 2014. *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Edited by O. Edenhofer, et al. Cambridge, United Kingdom and New York, NY, USA: Cambride University Press.

ISO 14040 International Standard. 2006. Environmental management - Life cycle assessment - Principles and framework. Geneva, Switzerland: International Organization for Standardization.

Jelinski, L. W., T. E. Graedel, R. A. Laudise, D. W. Mccall, and C. K. N. Patel. 1992. Industrial Ecology - Concepts and Approaches. *Proceedings of the National Academy of Sciences of the United States of America* 89(3): 793-797.

Joshi, S. 2000. Product environmental Life-Cycle Assessment using Input-Output Techniques. *Journal of Industrial Ecology* 3(2-3): 95-120.

Kennedy, M. C. and A. O'Hagan. 2001. Bayesian calibration of computer models. *Journal of the Royal Statistical Society Series B-Statistical Methodology* 63: 425-450.

Kim, H. C., V. Fthenakis, J.-K. Choi, and D. E. Turney. 2012. Life Cycle Greenhouse Gas Emissions of Thin-film Photovoltaic Electricity Generation. *Journal of Industrial Ecology* 16: S110-S121.

Kleijn, R., E. van der Voet, G. J. Kramer, L. van Oers, and C. van der Giesen. 2011. Metal requirements of low-carbon power generation. *Energy* 36(9): 5640-5648.

Klöpffer, W. 2006. The role of SETAC in the development of LCA. *International Journal of Life Cycle Assessment* 11: 116-122.

Liu, Z., D. Guan, W. Wei, S. J. Davis, P. Ciais, J. Bai, S. Peng, Q. Zhang, K. Hubacek, G. Marland, R. J. Andres, D. Crawford-Brown, J. Lin, H. Zhao, C. Hong, T. A. Boden, K. Feng, G. P. Peters, F. Xi, J. Liu, Y. Li, Y. Zhao, N. Zeng, and K. He. 2015. Reduced carbon emission estimates from fossil fuel combustion and cement production in China. *Nature* 524(7565): 335-338.

Majeau-Bettez, G., A. H. Stromman, and E. G. Hertwich. 2011. Evaluation of Process- and Input-Output-based Life Cycle Inventory Data with Regard to Truncation and Aggregation Issues. *Environmental Science & Technology* 45(23): 10170-10177.

Majeau-Bettez, G., R. Wood, and A. H. Strømman. 2014. Unified Theory of Allocations and Constructs in Life Cycle Assessment and Input-Output Analysis. *Journal of Industrial Ecology* 18(5): 747-770.

Majeau-Bettez, G., R. Wood, E. G. Hertwich, and A. H. Strømman. 2015. When Do Allocations and Constructs Respect Material, Energy, Financial, and

Production Balances in LCA and EEIO? *Journal of Industrial Ecology* 10.1111/jiec.12273.
Masanet, E., Y. Chang, A. R. Gopal, P. Larsen, W. R. Morrow, R. Sathre, A. Shehabi, and P. Zhai. 2013. Life-Cycle Assessment of Electric Power Systems. *Annual Review of Environment and Resources* 38(1): 107-136.
McGlade, C. and P. Ekins. 2015. The geographical distribution of fossil fuels unused when limiting global warming to 2 degrees C. *Nature* 517(7533): 187-U143.
Meinshausen, M., N. Meinshausen, W. Hare, S. C. B. Raper, K. Frieler, R. Knutti, D. J. Frame, and M. R. Allen. 2009. Greenhouse-gas emission targets for limiting global warming to 2 degrees C. *Nature* 458(7242): 1158-U1196.
Moriguchi, Y., Y. Kondo, and H. Shimizu. 1993. Analysing the life cycle impacts of cars: The case of CO2. *Industry and Environment* 16(1/2): 42-45.
Mutel, C. L., S. Pfister, and S. Hellweg. 2012. GIS-Based Regionalized Life Cycle Assessment: How Big Is Small Enough? Methodology and Case Study of Electricity Generation. *Environmental Science & Technology* 46(2): 1096-1103.
Nakhamkin, M. 2008. *150-300-400 MW CAES Plants Based on Various Combustion Turbines*. http://www.espcinc.com/library/150-300-400_MW_CAES_Brochure.pdf:
NETL, National Energy Technology Laboratory. 2010. *Methane Emissions from Mining Illinois Basin Coals*. DOE/NETL-2010/1445. Pittsburgh PA:
NETL, National Energy Technology Laboratory. 2014 *Life Cycle Analysis of natural gas extraction and power generation*. DOE/NETL-2014/1646. Pittsburgh, PA:
O'Donoughue, P. R., G. A. Heath, S. L. Dolan, and M. Vorum. 2014. Life Cycle Greenhouse Gas Emissions of Electricity Generated from Conventionally Produced Natural Gas. *Journal of Industrial Ecology* 18(1): 125-144.
Pauliuk, S., G. Majeau-Bettez, C. L. Mutel, B. Steubing, and K. Stadler. 2015. Lifting Industrial Ecology Modeling to a New Level of Quality and Transparency: A Call for More Transparent Publications and a Collaborative Open Source Software Framework. *Journal of Industrial Ecology* 10.1111/jiec.12316.
Pehnt, M., M. Oeser, and D. J. Swider. 2008. Consequential environmental system analysis of expected offshore wind electricity production in Germany. *Energy* 33(5): 747-759.
Peters, G. P. 2007. Efficient algorithms for life cycle assessment, input-output analysis, and Monte-Carlo analysis. *International Journal of Life Cycle Assessment* 12(6): 373-380.
Peters, G. P. and E. G. Hertwich. 2006. A comment on "Functions, commodities and environmental impacts in an ecological-economic model". *Ecological Economics* 59(1): 1-6.
Prado-Lopez, V., B. A. Wender, T. P. Seager, L. Laurin, M. Chester, and E. Arslan. 2015. Tradeoff Evaluation Improves Comparative Life Cycle Assessment: A Photovoltaic Case Study. *Journal of Industrial Ecology* 10.1111/jiec.12292.
Pré, Consultants BV. 2015. SimaPro. http://simapro.com/. Accessed 24 Sep 2015.
Raichur, V., D. S. Callaway, and S. J. Skerlos. 2015. Estimating Emissions from Electricity Generation Using Electricity Dispatch Models: The Importance of System Operating Constraints. *Journal of Industrial Ecology* 10.1111/jiec.12276.

Ravikumar, D., T. P. Seager, M. V. Chester, and M. P. Fraser. 2014. Intertemporal Cumulative Radiative Forcing Effects of Photovoltaic Deployments. *Environmental Science & Technology* 48(17): 10010-10018.

Renner, G. T. 1947. Geographyof Industrial Localization. *Economic Geography* 23(3): 167-189.

Rockström, J., W. Steffen, K. Noone, A. Persson, F. S. Chapin, E. F. Lambin, T. M. Lenton, M. Scheffer, C. Folke, H. J. Schellnhuber, B. Nykvist, C. A. de Wit, T. Hughes, S. van der Leeuw, H. Rodhe, S. Sorlin, P. K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R. W. Corell, V. J. Fabry, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen, and J. A. Foley. 2009. A safe operating space for humanity. *Nature* 461(7263): 472-475.

Saghafi, A. 2012. A Tier 3 method to estimate fugitive gas emissions from surface coal mining. *International Journal of Coal Geology* 100: 14-25.

Sanchez, D. L., J. H. Nelson, J. Johnston, A. Mileva, and D. M. Kammen. 2015. Biomass enables the transition to a carbon-negative power system across western North America. *Nature Climate Change* 5(3): 230-234.

Singh, B., A. H. Strømman, and E. G. Hertwich. 2011. Comparative life cycle environmental assessment of CCS technologies. *International Journal of Greenhouse Gas Control* 5(4): 911-921.

Singh, B., G. Guest, R. M. Bright, and A. H. Stromman. 2014. Life Cycle Assessment of Electric and Fuel Cell Vehicle Transport Based on Forest Biomass. *Journal of Industrial Ecology* 18(2): 176-186.

Singh, B., E. A. Bouman, A. H. Strømman, and E. G. Hertwich. 2015. Material use for electricity generation with carbon dioxide capture and storage: Extending life cycle analysis indices for material accounting. *Resources, Conservation and Recycling* 100(0): 49-57.

Skar, C., G. Doorman, and A. Tomasgard. 2014. The future European power system under a climate policy regime. Paper presented at Energy Conference (ENERGYCON), 2014 IEEE International, 13-16 May 2014.

Soares, S. R., L. Toffoletto, and L. Deschenes. 2006. Development of weighting factors in the context of LCIA. *Journal of Cleaner Production* 14(6-7): 649-660.

Stoft, S. 2002. *Power System Economics. Designing Markets for Electricity*. New York, NY, USA: John Wiley & Sons, Inc.

Stromman, A. H., G. P. Peters, and E. G. Hertwich. 2009. Approaches to correct for double counting in tiered hybrid life cycle inventories. *Journal of Cleaner Production* 17(2): 248-254.

Su, S., J. Y. Han, J. Y. Wu, H. J. Li, R. Worrall, H. Guo, X. Sun, and W. G. Liu. 2011. Fugitive coal mine methane emissions at five mining areas in China. *Atmospheric Environment* 45(13): 2220-2232.

Suh, S. 2004. Functions, commodities and environmental impacts in an ecological–economic model. *Ecological Economics* 48(4): 451-467.

Suh, S. 2006. Reply: Downstream cut-offs in integrated hybrid life-cycle assessment. *Ecological Economics* 59(1): 7-12.

Suh, S. and G. Huppes. 2005. Methods for Life Cycle Inventory of a product. *Journal of Cleaner Production* 13(7): 687-697.

Suh, S. and Y. Yang. 2014. On the uncanny capabilities of consequential LCA. *International Journal of Life Cycle Assessment* 19(6): 1179-1184.

Sørstrøm, S. E. 2001. *Konsekvensutredning for Svea Nord*. Store Norske Spitsbergen Kulkompani.

Taylor, K. E., R. J. Stouffer, and G. A. Meehl. 2012. An Overview of Cmip5 and the Experiment Design. *Bulletin of the American Meteorological Society* 93(4): 485-498.

Turconi, R., A. Boldrin, and T. Astrup. 2013. Life cycle assessment (LCA) of electricity generation technologies: Overview, comparability and limitations. *Renewable & Sustainable Energy Reviews* 28: 555-565.

UNFCCC, United Nations Framework Convention on Climate Change. 2012. National inventory submissions 2012. http://unfccc.int/national_reports/annex_i_ghg_inventories/national_inventories_submissions/items/6598.php. Accessed 17 Sep 2015.

Unruh, G. C. 2000. Understanding carbon lock-in. *Energy Policy* 28(12): 817-830.

van Berkel, R., E. Willems, and M. Lafleur. 1997. Development of an industrial ecology toolbox for the introduction of industrial ecology in enterprises-I. *Journal of Cleaner Production* 5(1-2): 11-25.

van Vuuren, D., J. Edmonds, M. Kainuma, K. Riahi, A. Thomson, K. Hibbard, G. Hurtt, T. Kram, V. Krey, J.-F. Lamarque, T. Masui, M. Meinshausen, N. Nakicenovic, S. Smith, and S. Rose. 2011. The representative concentration pathways: an overview. *Climatic Change* 109(1-2): 5-31.

Warner, E. S. and G. A. Heath. 2012. Life Cycle Greenhouse Gas Emissions of Nuclear Electricity Generation. *Journal of Industrial Ecology* 16: S73-S92.

Weber, C. L. and C. Clavin. 2012. Life Cycle Carbon Footprint of Shale Gas: Review of Evidence and Implications. *Environmental Science & Technology* 46(11): 5688-5695.

Weidema, B. P. 2014. Comparing Three Life Cycle Impact Assessment Methods from an Endpoint Perspective. *Journal of Industrial Ecology* 10.1111/jiec.12162.

Weidema, B. P. and M. S. Wesnæs. 1996. Data quality management for life cycle inventories—an example of using data quality indicators. *Journal of Cleaner Production* 4(3–4): 167-174.

Whitaker, M., G. A. Heath, P. O'Donoughue, and M. Vorum. 2012. Life cycle greenhouse gas emissions of coal-fired electricity generation: Systematic review and harmonization (vol 16, pg S53, 2012). *Journal of Industrial Ecology* 16(6): 966-966.

White, R. M. 1994. *The greening of industrial ecosystems*. Edited by B. R. Allenby and D. J. Richard. Washington D.C.: National Academy Press.

Yellishetty, M., P. G. Ranjith, A. Tharumarajah, and S. Bhosale. 2009. Life cycle assessment in the minerals and metals sector: a critical review of selected issues and challenges. *International Journal of Life Cycle Assessment* 14(3): 257-267.

Zamagni, A., J. Guinee, R. Heijungs, P. Masoni, and A. Raggi. 2012. Lights and shadows in consequential LCA. *International Journal of Life Cycle Assessment* 17(7): 904-918.

Appendix A: Paper I

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Multiregional environmental comparison of fossil fuel power generation—Assessment of the contribution of fugitive emissions from conventional and unconventional fossil resources

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ABSTRACT

In this paper we investigate the influence of fugitive methane emissions from coal, natural gas, and shale gas extraction on the greenhouse gas (GHG) impacts of fossil fuel power generation through its life cycle. A multiregional hybridized life cycle assessment (LCA) model is used to evaluate several electricity generation technologies with and without carbon dioxide capture and storage. Based on data from the UNFCCC and other literature sources, it is shown that methane emissions from fossil fuel production vary more widely than commonly acknowledged in the LCA literature. This high variability, together with regional disparity in methane emissions, points to the existence of both significant uncertainty and natural variability. The results indicate that the impact of fugitive methane emissions can be significant, ranging from 3% to 56% of total impacts depending on type of technology and region. Total GHG emissions, in CO_2-eq./kWh, vary considerably according to the region of the power plant, plant type, and the choice of associated fugitive methane emissions, with values as low as 0.08 kg CO_2-eq./kWh and as high as 1.52 kg CO_2-eq./kWh. The variability indicates significant opportunities for controlling methane emissions from fuel chains.

1. Introduction

With the increasing interest in power generation from unconventional fossil fuel resources, such as shale gas, and the large push for gas fired power plants as a clean form of electricity production (Stephenson et al., 2012), a more complete quantification of the (potential) environmental impacts of fossil fuel power generation life cycle is needed. Though the environmental impacts of the operation of most power generation technologies are relatively well described and quantified in life cycle assessment (LCA) literature (Corsten et al., 2013; Heath et al., 2014; O'Donoughue et al., 2014; Whitaker et al., 2012), we argue here that attention should also be directed towards upstream processes, such as the extraction and transport of fossil fuel resources (Alvarez et al., 2012; Burnham et al., 2012; Weber and Clavin, 2012). The fuel supply is especially important when carbon dioxide capture and storage (CCS) technology is applied to reduce the greenhouse gas emissions of the power plant itself, a step which increases fuel consumption due to the inherent energy efficiency penalty related to the carbon dioxide capture and compression processes.

One of the major greenhouse gases (GHGs) emitted in natural gas and coal production is methane. As a major constituent of natural gas, methane emissions occur at all points during the natural gas extraction process: well drilling and completion, well operation, e.g. in the form of purges and vents, and through leakages of the entire natural gas infrastructure, e.g., at intermediate compressor and redistribution stations of the pipeline (Burnham et al., 2012). Coal bed methane is formed from bacterial degradation of coal and biomass residuals, and thermally through devolatilisation within the coalification process of organic matter (Moore, 2012). It is released during coal extraction and removal of overburden. Methane emissions from fossil fuel origin are estimated to represent about 30% of the world anthropogenic methane emissions, although both fossil emissions and total anthropogenic emissions are quite uncertain (Kirschke et al., 2013).

A range of life cycle assessments (LCAs) of fossil fuel power generation with and without CCS has been published previously (Jaramillo et al., 2007; Koornneef et al., 2008; NETL, 2010b,c,d,e; Odeh and Cockerill, 2008; Singh et al., 2011a; Zapp et al., 2012). Most studies were thoroughly reviewed in the papers by Whitaker

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et al. (2012), O'Donoughue et al. (2014), Heath et al. (2014), and Corsten et al. (2013). Whitaker et al. (2012) present a review and harmonization of LCA greenhouse gas emission results for coal based electricity generation. Coal methane emissions are discussed, and an interquartile range of the reviewed studies of 54–73 g CO_2-eq/kWh is presented (median 63 g CO_2-eq/kWh). O'Donoughue et al. (2014) review and harmonize LCA greenhouse gas emission results for conventional gas based electricity generation. Heath et al. (2014) harmonize shale gas life cycle emissions. Methane leakage is discussed and ranges from 0.2% to 6% of natural gas production in the reviewed studies. Corsten et al. (2013) review the LCAs of both coal and natural gas based electricity generation in combination with CCS. They conclude that the upstream emissions of natural gas lead to large impacts on the overall GHG emissions, to the extent that electricity generated by a natural gas combined cycle power plant with CCS appears to have associated GHG emissions of the same order of magnitude as pulverized coal generated electricity with CCS.

Several recent studies focus on fugitive methane emissions from conventional and unconventional fossil fuel production. Weber and Clavin (2012) perform a Monte Carlo analysis based on six previous studies for natural gas from conventional and unconventional sources. Burnham et al. (2012) compare results for emissions related to coal and natural gas, shale gas and petroleum. Both studies conclude that upstream methane leakage and venting can reduce significantly the life cycle benefit from gas compared to coal, and that gas related emissions from conventional or shale production are statistically indistinguishable in a life cycle perspective. Laurenzi and Jersey (2013) study GHG emissions and water consumption of Marcellus shale gas production, but indicate that for certain GHG emissions EPA emission factors are used. They find that the estimated ultimate recovery of shale wells is one of the major determinants in the life cycle GHG emissions of shale gas electricity generation.

Though there are differences between the LCA studies of power plants with and without CCS in the literature, relatively little attention has been paid to fugitive emissions. These are mainly included by application of an emission factor and sometimes discussed as a subject of sensitivity analysis. In addition, most studies have a limited regional scope, evaluating power plants in Europe or the United States, with the shale gas literature focusing almost solely on the United States. This leads to the questions to what extent data are available with respect to fugitive methane emissions for both coal and natural gas, how they vary regionally, and consequentially what that implies for the environmental performance of fossil fuel power generation with and without CCS.

The aim of this paper is to make an inventory of the ranges of fugitive methane emissions available in the literature and assess the consequences these emissions have on the life cycle GHG impacts of fossil fuel power generation. We focus on fugitive methane emissions of coal mining, conventional natural gas production and shale gas production. The hybridized multiregional life cycle assessment model THEMIS (Technology Hybridized Environmental-economic Model with Integrated Scenarios) is used (Hertwich et al., 2014), in combination with a set of life cycle inventories for state-of-the-art fossil fuel power plants, both with and without CCS facilities. We allow for regional variation of fugitive emissions in order to increase understanding of the environmental consequences of implementation of fossil fuel power generation in different regions.

2. Methods

In this section we discuss the approach followed to assemble the fugitive emission datasets with special focus on the data reported in UNFCCC. We continue with a description of the HLCA model employed. The system description for the HLCA and life cycle inventories used are described separately in Section 3 of this paper.

2.1. Dataset assembly fugitive emissions

Three datasets were compiled containing a total of 227 entries for coal fugitive emissions, 34 entries for conventional gas fugitive emissions and 19 entries for shale gas emissions, based on peer reviewed published literature as well as data reported as part of the United Nations Framework Convention on Climate Change (UNFCCC). The UNFCCC was established in 1992 at the United Nations Conference on Environment and Development in Rio de Janeiro. The treaty has the objective to achieve '. . .*stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system*. . .' (United Nations, 1992). Annex I countries that have ratified the convention, report national greenhouse gas inventories yearly in the form of a national inventory report (NIR) and the common reporting format (CRF). The NIRs contain detailed information for each country and the CRF is an electronically submitted series of standardized data tables for all greenhouse gas emissions per sector. According to the guidelines governing the reporting on annual inventories, the estimates of emissions should be comparable among parties. In order to do so, countries have to follow the IPCC guidelines (IPCC, 2006) to estimate and report on anthropogenic emissions, but are free to use the different methods included in those guidelines (UNFCCC, 2004). Though data should be comparable between countries, there are different levels of uncertainty related to the UNFCCC data, which are related to the different calculation approaches accepted in the IPCC guidelines. Countries can report data using a tier 1 approach. In this approach, associated with the highest level of uncertainty, total emissions are calculated using a global average range of emissions factors and country-specific activity data. In the tier 2 approach, emissions are calculated using country or basin specific emissions factors. In the tier 3 approach, associated with the lowest level of uncertainty, direct measurements on a mine-specific basis are used (IPCC, 2006). Though not reported in the tables of the CRF, the NIRs contain information about the approaches used by Annex I countries (commonly mixes between tiers 1, 2, and 3) in reporting emissions data.

In this paper, we used the data provided by the Annex I countries in Table 1.B.1 and 1.B.2 of the CRF, that describe the fugitive emissions from solid fuels (1.B.1) and oil, natural gas and other sources (1.B.2) (UNFCCC, 2012). We selected for each country the average, minimum and maximum emissions of the time series from the starting year of reporting (usually 1990, though there are variations between countries) until 2010. These country level data were subsequently aggregated to generate a list of regional estimates of methane emissions related to coal production and conventional natural gas production. The regions correspond to the regional division of our HLCA model, which is described in Section 2.2.

In this study, values larger than 1.5 times the global interquartile range above the (global) 3rd quartile were considered outliers and were removed from the database. This was the case for natural gas data reported by Ukraine and Greece (respectively 1025 and 837 g CH_4/m^3 natural gas) and some of the coal data for Russia and France. Such high numbers may be due to the application of too uncertain emissions factors in the tier 1 method and possibly aggregation of fugitive emissions related to the natural gas transportation infrastructure in the UNFCCC common reporting format.

Because the United States is the only country with significant past shale gas production and because there is no distinction in the UNFCCC natural gas data regarding the source (conventional or shale) of methane emissions, we assumed that UNFCCC natural gas

Table 1
Regional coverage of datasets investigated.

Reference	Coal	Conventional gas	Shale gas	Regions[a]
UNFCCC (2012)	X	X		RER;US;PAC;EIT
Burnham et al. (2012)	X	X	X	US
Weber and Clavin (2012)		X	X	US
Howarth et al. (2011)		X	X	US
Sørstrøm (2001)	X			RER;US;EIT
Su et al. (2011)	X			CN
Bibler et al. (1998)	X			CN
EPA (2006)	X			RER;US; EIT
Saghafi (2012)	X			PAC
NETL (2010f)	X			US
NETL (2014)		X	X	US

[a] Region abbreviations are: CN = China, RER = OECD Europe, US = OECD North America, PAC = OECD Pacific, EIT = Economies in Transition

emissions data are only relevant for the conventional natural gas system.

In addition to official emissions reports scientific literature sources were consulted. Coal mining, conventional natural gas extraction, and shale gas extraction are described by Burnham et al. (2012). Shale gas is included in (Howarth et al., 2011; NETL, 2014; Weber and Clavin, 2012). A set of emissions factors for coal mines was found for the regions China, OECD Pacific and Economies in Transition (mainly Russia) (Bibler et al., 1998; EPA, 2006; NETL, 2010f; Saghafi, 2012; Su et al., 2011; Sørstrøm, 2001), thus generating at least one dataset for five different regions in the HLCA model. Table 1 shows the regional coverage of the three datasets compiled based on the references consulted. The total number of data points per region and source is presented in Table ST1 of the supporting information.

2.2. HLCA model

A multi-regional integrated hybrid life cycle assessment (HLCA) model was employed to model the potential environmental impacts (Hertwich et al., 2014). We modelled a traditional process based Life Cycle Assessment and complemented this with economic data where these were available. The model was set-up as a tiered hybrid model, in which it is possible to select for each unit process background data from both a physical inventory, ecoinvent 2.2 (Dones et al., 2007), and an environmentally extended Input-Output database EXIOBASE (Tukker et al., 2013). In the THEMIS model, EXIOBASE is aggregated to nine regions from its original regional classification, but incorporates a disaggregated electricity sector (Hertwich et al., 2014). Potential environmental impacts were calculated on a per-kWh electricity produced functional unit basis. For the LCA we took a cradle-to-gate approach.

As methane is an important greenhouse gas, we evaluated GHG emissions using Global Warming Potentials (GWPs) over a 100-year time horizon. For each of the emission factors found in the literature the appropriate stressors were adapted and the LCA model was run, which generated a range of model outcomes for the climate change impact associated with the fossil electricity generation. The ecoinvent database contains nine unique processes that cover natural gas extraction and 10 processes for the extraction of hard coal. A shale gas extraction process did not exist in the database, and therefore an inventory was built based on data from the Argonne National Laboratory (Clark et al., 2011). All modelling was performed in Matlab in combination with an Excel interface for data input.

The life cycle inventory data are based on state-of-the-art power plants described by several reports of the National Energy Technology Laboratory in the US. These studies present detailed cost economic modelling of power plants and life cycle inventories (NETL, 2010a,b,c,d,e), thus providing a suitable starting point for hybrid life cycle assessment. Where data were not sufficient, or too specific for a generic power plant, peer reviewed literature was consulted to provide input data (Koornneef et al., 2008; Singh et al., 2011a; Veltman et al., 2010).

3. Life cycle inventory

Four different types of electricity production technologies were modelled. The investigated technologies are:

(i) subcritical pulverized coal fired power (Sub-PC)
(ii) supercritical pulverized coal fired power (SCPC)
(iii) integrated gasification combined cycle (IGCC)
(iv) natural gas combined cycle (NGCC)

Out of these technologies, three are connected to a post-combustion CO_2 capture process (using amine as solvent) and one is connected to a pre-combustion CO_2 capture process (using a solvent consisting of dimethyl ethers and polyethylene glycol). Key characteristics of these technologies are described in Table 2. We evaluate the power plants on a cradle-to-gate perspective. Electricity transport and distribution to the end users is outside the scope of the study. Each life cycle inventory is set up according to the following structure: fossil fuel extraction, fossil fuel transport, power plant operation and a separate foreground process for power plant infrastructure. For the inventories in which carbon capture and storage is included, the following foreground processes are added: CO_2 capture and compression, CO_2 transport by pipeline, and the CO_2 injection well. The key foreground processes are shortly discussed in the following sections. Information regarding specific emissions and the efficiencies of emissions reduction measures implemented with each power plant is given in Table ST2 in the supporting information.

The following sections describe our base inventory for the four investigated power plant technologies. As the purpose of this paper is to show how varying emissions upstream can influence the LCA results related to power generation we do not change the efficiency of the power plants between regions. However, as our base inventory (presented in Tables ST5–ST16) is based on fuels with very specific energy and carbon density, we assume a regional specific lower heating value (LHV) for the fuel used in order to adapt the fuel requirement and direct emissions of power plant operation for each region. The scaling factors we developed to adapt these flows in the base inventory are presented in Tables ST3 and ST4 of the supporting information. The regional specific LHV is used to calculate the fossil fuel input for the power plant in each region. Direct emissions of power plant operation are scaled with both regional specific LHVs and carbon content. To that extent, we assembled a set of coal carbon content (CC) and LHV pairs (in the range of 18–31 MJ/kg coal), that were used in previous LCAs and express CC as function of LHV (Whitaker et al., 2012). In the specified LHV

Table 2
Key power plant characteristics (NETL, 2010a).

	Unit	Sub-PC	SCPC	IGCC	NGCC
Net power output without CCS	MW	550	550	629	555
Net power output with CCS	MW	550[a]	550[a]	497	474
Capacity factor	%	85	85	80	85
Net plant efficiency without CCS	%	38.2	40.7	43.6	55.6
Net plant efficiency with CCS	%	27.2	29.4	32.3	47.4
Fuel requirements	kg/kWh	0.361	0.338	0.315	0.187 (m^3/kWh)
Fuel requirements with CCS	kg/kWh	0.507	0.467	0.425	0.219 (m^3/kWh)
CO_2 emissions from power plant	g/kWh	856	802	723	365
CO_2 emissions from power plant with CCS	g/kWh	120	111	109	42.6
MEA consumption	kg/tonne CO_2	2.15	2.15	0.09 (dimethyl ether)	2.15
CO_2 capture efficiency	%	90	90	90	90
Lifetime	years	30	30	30	30

[a] The nominal net output for the Sub-PC and SCPC cases was maintained at 550 MW for the cases with CCS. This is done by increasing the boiler and turbine/generator sizes to account for a larger auxiliary load due to the carbon capture process. For the IGCC and NGCC cases, the plant size was kept constant, leading to a lower net power output.

range we assumed that this function behaves linearly for all practical purposes. The scaling factor for direct power plant emissions was calculated based on the relative change of the ratio between calculated CC and regionally specified LHV. Since the variation in the LHV of natural gas used in the model is relatively low, we have chosen to use the same scaling factor for both natural gas inputs and emissions.

3.1. Fossil fuel extraction

Three types of extraction processes are modelled in this paper: coal mining, conventional natural gas extraction, and shale gas extraction. For coal and conventional natural gas the ecoinvent processes *hard coal*, *at mine* and *natural gas*, *at production* are used, with the exception of making the fugitive methane emissions in these processes a model variable. Please note that, for coal, we do not explicitly distinguish between underground and surface coal mining processes, but use the underground/surface mine ratio in the ecoinvent database.

A shale gas extraction process was modelled based on data published by the Argonne National Laboratory (Clark et al., 2011). A well production over a lifetime of 30 years of 98 million cubic metres was assumed, though it should be noted that much shorter lifetimes have been reported (O'Sullivan and Paltsev, 2012). Material requirements for the drilling and construction of the well pads are taken as the non-weighted average of four shale gas plays in the United States (namely, Barnett, Marcellus, Fayetteville, and Haynesville). The fracking fluid is a mixture of water and sand with a range of organic and inorganic chemicals such as methanol, hydrogen chloride, formaldehyde, sodium hydroxide and ethylene glycol. The inventory for the fracking fluid is given in Table ST5 of the supporting information.

Electricity and diesel fuel consumption for well operation are taken as an average of four wells described by Clark et al. (2011). Within our model, the emissions associated with well completion and well workovers are not explicitly stated, but are part of the well operation process, as many sources report well completion in percentage of natural gas during production. The methane emissions for well completion and workovers are reported to be 417 tonnes of methane per well over its life cycle (Clark et al., 2011). Table ST6 in the supporting information shows the required material inputs and methane emissions associated with the construction of a shale gas well as modelled in this study and Table ST7 shows the inventory for shale well operation.

3.2. Fossil fuel transport

In this study, the coal fired power plants are assumed to use the same coal transport unit process. Coal is transported by rail over a distance of 330 km from the excavation site to the power plant (NETL, 2010e). The material requirements for the trains are included in the inventory, as well as diesel required for transport. The rails themselves are assumed to be constructed and available and are not included in the inventory. During coal extraction and transport it is assumed that no coal is lost. The coal transport inventory is presented in Table ST8 of the supporting information.

Gas is assumed to be transported 1000 km by pipeline, connecting an offshore natural gas extraction site and the power plant location (ecoinvent process *transport*, *natural gas*, *pipeline*, *long-distance*). Although the shale gas wells are land based and it would be expected that the transport distance is shorter, it was chosen to keep the pipeline length constant, in order to make inventories more comparable. Methane leakage during transport is assumed at 0.026% of transported natural gas per 1000 km based on ecoinvent (Faist Emmenegger et al., 2007).

For Russia, the ecoinvent leakage rate is considerably higher at 0.23% per 1000 km (1.4% for a transport distance of 6000 km) (Faist Emmenegger et al., 2007). Leakage rates for transmission and distribution of 0.67% (0.29–1.05%) to 1.5% (0.8–2.2%) are reported for the US, but a specific transport distance is not reported (Burnham et al., 2012; Weber and Clavin, 2012). To study the increase in contribution of methane to the life cycle impacts, the natural gas transport process was updated with the values for the EIT (0.23%) and the US (0.67%). We report the influence of different natural gas pipeline fugitive emissions rates in Section 4.3.

3.3. Pulverized coal fired power plants

The baseline inventory includes both sub- and supercritical coal fired power technology (see Tables ST9 and ST10). Both coal fired power plants are based on designs from the National Energy Technology Laboratory (NETL, 2010a). Key plant characteristics are given in Table 2. Main inputs are taken from the ecoinvent background. The largest flows are hard coal fuel, limestone for the flue gas desulphurization unit, ammonia for the selective catalytic reduction of NO_x emissions and water for cooling duties. In addition, for the CCS processes, monoethanolamine (MEA), caustic soda, and activated carbon are also used. The treatment of waste generated by the power plants, is modelled following ecoinvent. Main emissions for the power plants without CCS are carbon dioxide, water vapor, particulate matter, sulfur dioxide and nitrogen oxides (NETL, 2010b,e). The flue gas desulphurization process in the coal fired power plants yields gypsum as an economic byproduct. In this paper, we take a conservative approach and no impacts are allocated to gypsum production. In power plants with CCS, ammonia and MEA emissions are also included. The

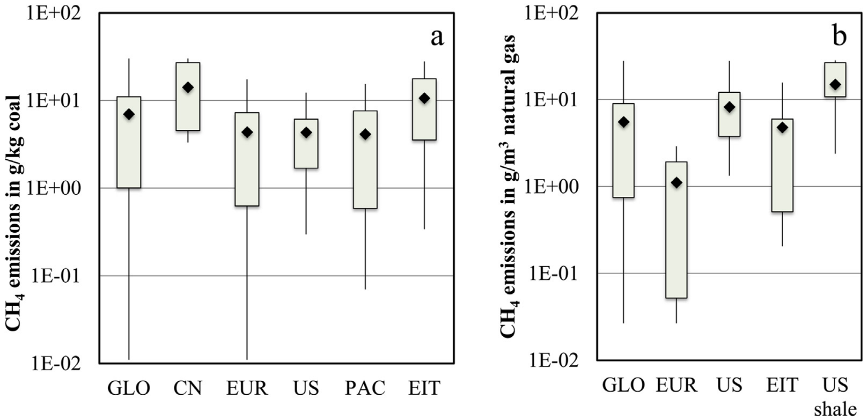

Fig. 1. (a, b) Reported fugitive methane emissions for the extraction of coal (a) and extraction of natural gas (b). GLO = global, CN = China, EUR = OECD Europe, US = OECD North America, PAC = OECD Pacific, EIT = Economies in Transition. N.B. Emissions associated with natural gas production from conventional and shale source is presented separately in columns US and US shale of panel (b).

CO_2 captured is transported in dense phase and is compressed on-site to 153 bar before transport. The electricity for compression is generated by the power plant and it is included in the energy penalty due to CO_2 capture. It is further assumed that no extra cleaning equipment is required and that dehydration during compression reduces the water content to at least 500 ppmv, making it suitable for transport. Power plant infrastructure, as well as chemicals that constitute minor inputs, are modelled using flows from the economic EXIOBASE background (see Tables ST12 and ST13).

3.4. Integrated gasification combined cycle

The integrated gasification combined cycle power plant is modelled based on the designs of NETL (NETL, 2010a). The key plant characteristics are given in Table 2. Main inputs are taken from the ecoinvent background (see Table ST8). Before combustion, coal is gasified producing a mixture of hydrogen and carbon monoxide. As noted before, the coal transport process is assumed to be the same as the transport process for the sub- and supercritical power plants. Due to its higher efficiency, the fuel requirements for the IGCC are somewhat lower than those for the pulverized coal power plants. Besides coal, the main inputs to plant operation are process water for cooling duties, catalyst for the COS hydrolysis unit (in the case of the IGCC without CCS) and activated carbon for the removal of mercury. In the case of IGCC with CCS, the water gas shift reactor also hydrolyses carbonyl sulphide (COS) into H_2S, hence no separate COS hydrolysis reactor is needed. A mixture of dimethyl ethers and polyethylene glycol is used as a physical solvent for both the IGCC plant with and without CCS and is used for mainly sulfur removal (single stage) or for both sulfur and CO_2 removal (dual stage). Though sulfur is a byproduct of the IGCC power plant, the same approach as with the gypsum production in the supercritical power plant is followed, thus impacts are not allocated with respect to sulfur. The solvent has a low vapor pressure, and spent solvent is therefore assumed to end up in the solid waste output of the power plant (Singh et al., 2011b). Main emissions for the IGCC power plant are carbon dioxide, water vapor, particulate matter, sulfur dioxide and nitrogen oxides. The CO_2 captured is compressed to 153 bar before transport. Power plant and CCS infrastructure, as well as chemicals that constitute minor inputs, are modelled using flows from the economic EXIOBASE background (see Tables ST15 and ST16).

3.5. Natural gas combined cycle

The natural gas plant is modelled as a combined cycle plant (NETL, 2010a). Besides natural gas, the main plant inputs are ammonia for the selective catalytic reduction (SCR) of NO_x, process water for cooling duties and chemicals such as the catalyst of the SCR unit. Inputs to the CO_2 capture process are activated carbon and MEA. Main emissions for the NGCC power plants are carbon dioxide, water vapor, ammonia, and nitrogen oxides (see Table ST9). The CO_2 captured is compressed to 153 bar before transport. Similar to the other inventories, infrastructure is modelled using the EXIOBASE economic background (see Tables ST15 and ST16).

3.6. CO_2 transport and storage

Captured carbon dioxide is assumed to be transported to an underground aquifer by pipeline. CO_2 is transported in dense phase over a transport distance of 150 km. As the inlet pressure was 153 bars, the pressure drop over the 150 km pipeline is small enough to prevent two-phase formation and therefore intermediate booster stations are not required. Following the approach by Singh et al. (2011a), pipeline inventory data are modelled after a high capacity offshore natural gas pipeline from ecoinvent (see Table ST13). Carbon dioxide leakage from the pipeline is assumed to be 0.01% of transported CO_2 (see Table ST14, Koornneef et al., 2008).

Captured CO_2 is injected in an aquifer at a depth of 1200 m. It is assumed that no booster compression is required at the wellhead, though this will be determined by site specific pressure conditions in the bottom well. The CO_2 injection rate per well is 9.4 kt CO_2 per day and is modelled as an offshore drilling well from ecoinvent (Singh et al., 2011a). In this study it is assumed that the reservoir is large enough to store the CO_2 over the lifetime of the power plant and that CO_2 is stored permanently (that is, there is no leakage from the reservoir).

4. Results

4.1. Dataset analysis

Fig. 1 shows the fugitive methane emissions within the data assembled. As can be seen for both coal and natural gas, fugitive emissions vary by orders of magnitude. The figure shows the outlier-adjusted minimum and maximum values for the different regions in the dataset (indicated by the lines), and the first and third quartile of the data (indicated by the box). In addition to the different regions, the global range is also presented. The regions China and Economies in Transition show clearly higher emissions associated with coal than the United States and Europe. There is a large spread in the European data as the result of some very low emissions (0.01 g CH_4/kg coal) reported in the UNFCCC data. Methane emissions from gas production in North America are larger than those in Europe and the Economies in Transition. This divergence raises the question to what extent higher reported emissions in the US are due to difference in practice and specific site conditions and to what degree it could be the result of more attention to the issue, as indicated by the relatively high attention for (US) fugitive emissions in scientific literature. The results also indicate that fugitive emissions associated with shale gas are on average higher than for conventional natural gas production. This can be due to the large uncertainty involved in the emissions associated with well completion and workover emissions. For example, these emissions are reported to be in the range of 0.006–2.75% of natural gas production (Burnham et al., 2012). Dataset analysis did not reveal an obvious distribution of the emissions factors in the UNFCCC data, even though a lognormal (Dones et al., 2007) and triangular distribution (Weber and Clavin, 2012) have been assumed previously for the purpose of Monte Carlo simulations.

It is important to note here that the large ranges of fugitive emissions shown are caused by both natural variation and uncertainty in the data. For example, differences in coal grade and rank between mines have an influence on the methane emissions included in the coal bed (Moore, 2012). Furthermore, surface mines are more likely to have been vented by natural processes and can therefore be expected to have lower associated fugitive emissions than underground mines. In addition, natural gas is captured from coal seams (coal bed methane) thereby reducing the potential fugitive emissions of to-be extracted coal (NETL, 2014). The range of emissions related to gas infrastructure is most likely a result of the inherent uncertainty involved in the quantification of emissions using the tier 1 and 2 methods.

4.2. Life cycle impact assessment

In this section, the results of the life cycle impact assessment are presented. Fig. 2 presents a boxplot of impact assessment results for the climate change impact category in g CO_2-eq per kWh for all technologies investigated and based on a global warming potential evaluated at a 100 year time horizon (Solomon et al., 2007). It is shown that the results vary considerably, with China and the Economies in Transition showing the highest impacts, as can be expected from the fugitive emissions range presented in Fig. 1. The full range of results for coal fired technology without CCS lies between 747 and 1303 g CO_2-eq./kWh of electricity produced. Not surprisingly, for the cases without CCS, natural gas power plant emissions are lower than coal fired power emissions, and lie between 367 and 533 kg CO_2-eq./kWh. For the coal fired power plants, the average contribution of methane emissions varies considerably between 4% in the OECD Pacific region and 15% in China. For the natural gas fired power plants this range is wider with the average contribution of methane ranging from 3% in Europe up to 16% for shale gas in the US.

Though there are large differences in the contribution of methane to GWP between regions, we see no significant difference in relative methane contribution for the three different coal technologies without CCS. It is important to note here that the difference between regions has a three-fold origin. The first one is the variation in the fugitive emissions rates between regions according to the data ranges shown in Fig. 1. The second is due to the introduction of the regional specific LHVs for coal and natural gas. In regions with relatively low LHV (e.g. China) the higher fuel requirements translate into a higher contribution of methane to the GWP. Thirdly, the regionalized background contained in THEMIS introduces some variation in regional GWPs. For example, the electricity mix used in the production of the diesel fuel used in the transport of coal varies between regions. In the case of fossil fuel power plants the contribution of the regionalized background is small, as most of the emissions are associated with the foreground processes.

Table 3
The contribution of methane to the life cycle GHG emissions of power production when including region-specific transport emissions.[a]

	EIT	US
NGCC	9% (8%)	16% (12%)
NGCC + CCS	34% (29%)	54% (45%)
NGCC shale	n.a.	20% (16%)
NGCC + CCS shale	n.a.	61% (56%)

[a] Values in brackets indicate the methane contribution with generic transport emissions previously used.

The inclusion of CCS decreases the environmental impacts of power plants considerably, with GHG results ranging from 128 to 747 g CO_2-eq./kWh for coal fired power plants. Results for natural gas plants lie between 75 and 250 g CO_2-eq./kWh. The average contribution of fugitive methane emissions after installing CCS technology ranges from 23% to 50% for coal and 19% to 56% for natural gas. Contrary to the cases without CCS, we can observe differences in the average contribution of methane emissions between technologies (for equal regions) since the direct emissions of the power plant become less dominant.

In the interest of comparability we have not included intra-regional changes in both LHV and efficiency of the power plant. An increase in power plant efficiency will shift the entire range of GWPs proportional to the decrease in fuel requirements. An increase in LHV would also result in lower fuel requirements, but most likely would affect direct power plant emissions much less due to the associated increase in carbon content. The opposite is valid for decreases in both efficiency and LHV. The above presented numbers show the importance of mitigation of methane emissions in the fossil fuel extraction process, as these emissions contribute largely to the emissions associated with fossil fuel power generation, especially for fuels with a relatively low LHV. It should be noted here that results for gas fired power plants and coal fired power plants partially overlap when carbon capture technology is installed, a conclusion also reached by for instance Corsten et al. (2013).

4.3. The influence of natural gas transport emissions

So far, we have explored only the fugitive emissions associated with the extraction of fossil fuels. However, emissions also occur in the transport of natural gas. As described before, the natural gas transport process was updated with new values for EIT (0.23%) and the US (0.67%). The results are presented in Table 3. We see a small increase for the EIT, even though emissions associated with transport are increased by an order of magnitude. Not surprisingly, the change is more apparent for North America, due to the high rate of emissions assumed to be associated with transport. However, it is not clear whether the 0.67% natural gas loss would be consistent with the pipeline length of 1000 km that is used in our model. Rather than estimating the contribution of emissions associated with natural gas transport, the purpose here is to show that emissions associated with transport have to become relatively high (as indicated by the US emissions rate) in order to become significant compared to fugitive emissions during extraction.

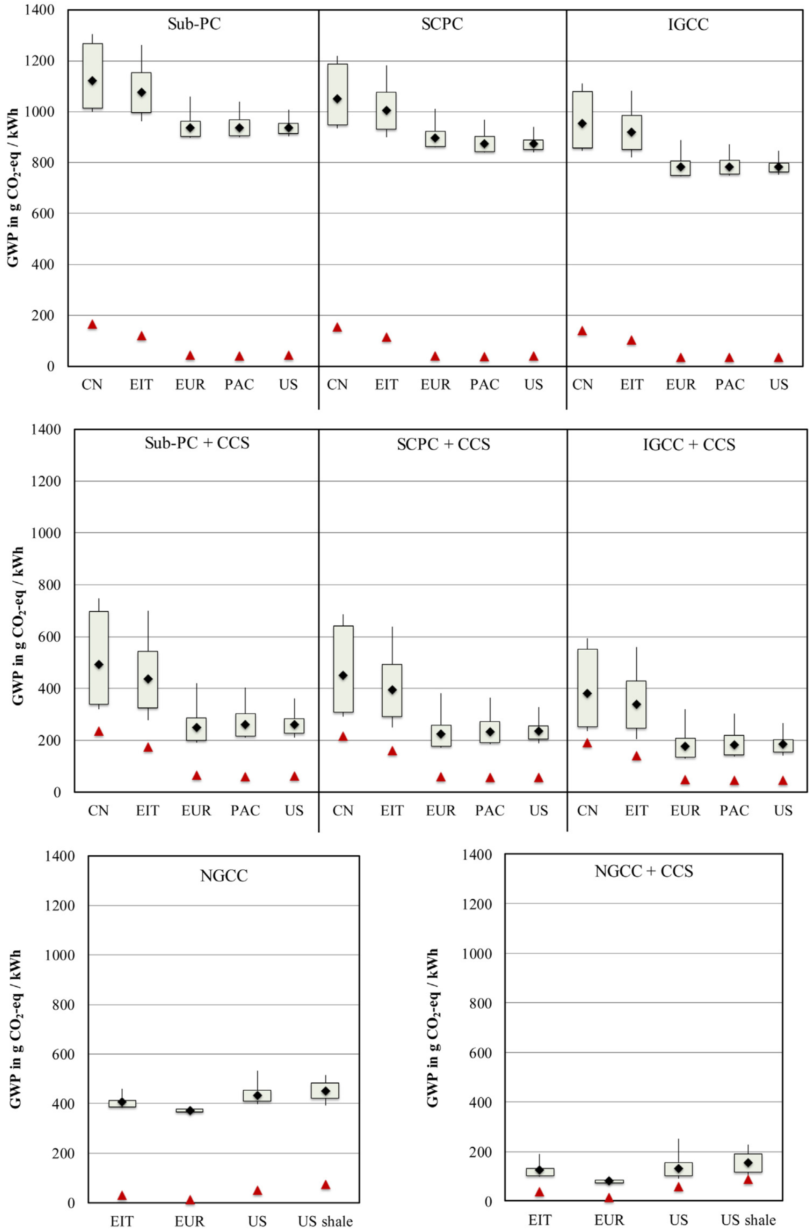

Fig. 2. Calculated Global Warming Potential per kWh energy produced in sub-, supercritical, integrated gasification coal fired power plants, and natural gas fired power plants for the year 2010. Results are based on different fugitive emissions during fossil fuel extraction. Sub-PC = subcritical pulverized coal, SCPC = supercritical pulverized coal, IGCC = integrated gasification combined cycle, NGCC = natural gas combined cycle. The plotted triangles indicate the average contribution of methane emissions to the impact assessment. The plotted diamonds indicate the average GWP.

5. Discussion

The direct comparison of LCA results between different studies is always hampered by differences in system boundaries, plant type investigated, and background database used. For example, Burnham et al. (2012) use an NGCC power plant efficiency of 47% and a supercritical coal power plant efficiency of 41.5% (compared to respectively 55.6% and 40.7% used in this paper). Modelling is performed with the GREET model, and not with ecoinvent. In this section we therefore compare qualitative results rather than quantitative results.

Burnham et al. (2012) have concluded that total upstream emissions can reduce the life-cycle benefit for natural gas compared to coal, which the current study also indicates. There is no agreement in the literature on the comparison on the magnitude of shale gas emissions compared to conventional natural gas emissions and appears to be strongly tied to the shale well lifetime and associated ultimate recovery (Laurenzi and Jersey, 2013; O'Sullivan and Paltsev, 2012). In our modelling we see on average a larger impact for US shale than for US conventional gas, but we would like to point out that the ranges overlap to a considerable extent. Both Weber and Clavin (2012) and Laurenzi and Jersey (2013) conclude that the relative difference in GWP between conventional and shale gas production is smaller than the uncertainty in either estimate. As gas is increasingly extracted from unconventional sources special attention to methane emissions could provide a significant mitigation opportunity.

While fossil fuel power plants with high GHG emissions are reported in the literature, these emissions are generally caused by a low efficiency of the power plants (Whitaker et al., 2012). Our results show that even modern power plants can have high life cycle GHG emissions due to fuel chain methane releases. They also show that fuel energy density and associated carbon content are an important parameter in determining fuel requirement, and hence the contribution of fugitive emissions, and direct emissions of power plant operation. It should be noted that the non-methane upstream contribution is in the order of 1–4%, mainly diesel combustion during operation of machinery and transport of coal, or carbon dioxide emissions associated with combustive processes during natural gas extraction and transport.

All impact results in this paper are reported using global warming potentials with a 100-year time horizon and a characterization factor for methane of 25 kg CO_2-eq/kg CH_4. In the latest round of IPCC reports, the characterization factor was updated to 34 kg CO_2-eq/kg CH_4. For GWPs evaluated over a 20-year time horizon the methane characterization factor is considerably larger at 86 CO_2-eq/kg CH_4 (Myhre et al., 2013). The methane characterization factors show that the contribution of methane to radiative forcing is significant, especially in the short term. Several authors have tried to capture this by developing alternative models such as Technology Warming Potential (Alvarez et al., 2012) and Time Adjusted Warming Potential (Kendall, 2012).

6. Conclusion

The aim of this paper was to provide a better understanding of methane emissions associated with the extraction of fossil fuels and assess their effect on the life cycle impacts of fossil fuel power generation. A set of life cycle inventories was assembled and combined with a dataset of fugitive methane emissions in a multiregional hybrid LCA model. The results of the dataset analysis reveals that fugitive emissions can vary by orders of magnitude, both inter- and intraregional. Our impact assessment results indicate that fuel chain methane emissions can constitute a substantial portion of the total emissions from fossil fuel power, and both their absolute magnitude and relative importance will increase with the deployment of CCS. In the most extreme cases, emissions from the fuel chain could be of equal importance to emissions from a power plant with CCS.

We see that methane emissions from fossil fuel production vary more widely than commonly acknowledged in the LCA literature, and that there are distinct regional disparities. By including the regionalization in our model we provide a more detailed picture of the contribution of fugitive methane emissions to the total life cycle impact. Coal methane emissions are more relevant for power plants in the regions China and Economies in Transition, with contributions over 40% for plants with CCS technology included, than for similar plants in Europe and North America. This is a result from higher fugitive emissions during extraction and the increased fuel requirements related to the use of fuel with a lower energy density. However, in the case of natural gas extraction, the contribution of fugitive emissions is significant for the North American region, with an average contribution that can exceed 50% for the plants with CCS technology. European conventional natural gas production appears to have the lowest amount of fugitive emissions associated. The inclusion of higher emissions associated with natural gas pipeline transport becomes only significant when gas leakage rates increase by at least an order of magnitude compared to leakage from the European grid, which was used as the default ecoinvent process.

The regional disparities may not reflect differences in geological factors, technologies, and practices employed. Most emissions estimates in both the UNFCCC data and literature are based on engineering calculations and not measurements, with only one paper utilizing actual measured shale gas production data. More measurements and an in-depth review of the engineering calculations are required to illuminate whether reported differences reflect actual variation in emissions or our uncertainty about them. A clear approach on how many of the data points are generated, i.e. using tier 1, 2, 3 or mixed methods, is preferable. In addition, most literature seems to focus on processes in the United States, but as this study shows, there is a need for detailed empirically determined emissions data in both North America and other regions, as the uncertainties related to the data reported under the UNFCCC common reporting format are not sufficiently quantified.

Given the large impact of methane emissions on LCA results we recommend practitioners to be aware of the sensitivity and to always perform a sensitivity analysis addressing uncertainty related to the upstream processes. Depending on timeframe and scope, there are examples of detailed inventories (NETL, 2014) in which fugitive emissions are addressed on a component specific level that could be adapted to specific conditions.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ijggc.2014.11.015.

References

Alvarez, R.A., Pacala, S.W., Winebrake, J.J., Chameides, W.L., Hamburg, S.P., 2012. Greater focus needed on methane leakage from natural gas infrastructure. Proc. Natl. Acad. Sci. U.S.A. 109, 6435–6440.

Bibler, C.J., Marshall, J.S., Pilcher, R.C., 1998. Status of worldwide coal mine methane emissions and use. Int. J. Coal Geol. 35, 283–310.

Burnham, A., Han, J., Clark, C.E., Wang, M., Dunn, J.B., Palou-Rivera, I., 2012. Life-cycle greenhouse gas emissions of shale gas, natural gas, coal, and petroleum. Environ. Sci. Technol. 46, 619–627.

Clark, C.E., Han, J., Burnham, A., Dunn, J.B., Wang, L., 2011. Life-Cycle Analysis of Shale Gas and Natural Gas. Argonne National Laboratory, Chicago, IL.

Corsten, M., Ramirez, A., Shen, L., Koornneef, J., Faaij, A., 2013. Environmental impact assessment of CCS chains – lessons learned and limitations from LCA literature. Int. J. Greenh. Gas Control 13, 59–71.

Dones, R., Bauer, C., Bolliger, R., Burger, B., Faist Emmenegger, M., Frischknecht, R., Heck, T., Jungbluth, N., Röder, A., 2007. Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries. Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

EPA, Environmental Protection Agency, 2006. Global Mitigation of Non-CO_2 greenhouse gases, Washington, DC.

Faist Emmenegger, M., Heck, T., Jungbluth, N., Tuchschmid, M., 2007. Erdgas. In: Dones, R., et al. (Eds.), Sachbilanzen von Energiesstemenen: Grundlagen für den ökologiscen Vergleich von Energiesystemenen un den Einbezug von Energiesystemenen in Ökobilanzen für die Schweiz. Paul Scherrer Institut Villigen. Swiss Centre for Life Cycle Inventories, Dübendorf, CH.

Heath, G.A., O'Donoughue, P., Arent, D.J., Bazilian, M., 2014. Harmonization of initial estimates of shale gas life cycle greenhouse gas emissions for electric power generation. Proc. Natl. Acad. Sci. U.S.A. 111, E3167–E3176.

Hertwich, E.G., Gibon, T., Bouman, E.A., Arvesen, A., Suh, S., Heath, G.A., Bergesen, J.D., Ramirez, A., Vega, M.I., Shi, L., 2014. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. Proc. Natl. Acad. Sci. U.S.A., http://dx.doi.org/10.1073/pnas.1312753111.

Howarth, R.W., Santoro, R., Ingraffea, A., 2011. Methane and the greenhouse-gas footprint of natural gas from shale formations. Clim. Change 106, 679–690.

IPCC, 2006. In: Eggleston, H.S., Buendia, L., Miwa, K., Ngara, T., Tanabe, K. (Eds.), 2006 IPCC Guidelines for National Greenhouse Gas Inventories, Japan.

Jaramillo, P., Griffin, W.M., Matthews, H.S., 2007. Comparative life-cycle air emissions of coal, domestic natural gas, LNG, and SNG for electricity generation. Environ. Sci. Technol. 41, 6290–6296.

Kendall, A., 2012. Time-adjusted global warming potentials for LCA and carbon footprints. Int. J. Life Cycle Assess. 17, 1042–1049.

Kirschke, S., Bousquet, P., Ciais, P., Saunois, M., Canadell, J.G., Dlugokencky, E.J., Bergamaschi, P., Bergmann, D., Blake, D.R., Bruhwiler, L., Cameron-Smith, P., Castaldi, S., Chevallier, F., Feng, L., Fraser, A., Heimann, M., Hodson, E.L., Houweling, S., Josse, B., Fraser, P.J., Krummel, P.B., Lamarque, J.-F., Langenfelds, R.L., Le Quere, C., Naik, V., O'Doherty, S., Palmer, P.I., Pison, I., Plummer, D., Poulter, B., Prinn, R.G., Rigby, M., Ringeval, B., Santini, M., Schmidt, M., Shindell, D.T., Simpson, I.J., Spahni, R., Steele, L.P., Strode, S.A., Sudo, K., Szopa, S., van der Werf, G.R., Voulgarakis, A., van Weele, M., Weiss, R.F., Williams, J.E., Zeng, G., 2013. Three decades of global methane sources and sinks. Nat. Geosci. 6, 813–823.

Koornneef, J., van Keulen, T., Faaij, A., Turkenburg, W., 2008. Life cycle assessment of a pulverized coal power plant with post-combustion capture, transport and storage of CO_2. Int. J. Greenh. Gas Control 2, 448–467.

Laurenzi, I.J., Jersey, G.R., 2013. Life cycle greenhouse gas emissions and freshwater consumption of marcellus shale gas. Environ. Sci. Technol. 47, 4896–4903.

Moore, T.A., 2012. Coalbed methane: a review. Int. J. Coal Geol. 101, 36–81.

Myhre, G., Shindell, D., Bréon, F.-M., Collins, W., Fuglestvedt, J., Huang, J., Koch, D., Lamarque, J.-F., Lee, D., Mendoza, B., Nakajima, T., Robock, A., Stephens, G., Takemura, T., Zhang, H., 2013. Anthropogenic and natural radiative forcing. In: Stocker, T.F., Qin, D., Plattner, G.-K., Tignor, M., Allen, S.K., Boschung, J., Nauels, A., Xia, Y., Bex, V., Midgley, P.M. (Eds.), Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA, pp. 659–740.

NETL, National Energy Technology Laboratory, 2010a. Cost and Performance Baseline for Fossil Energy Plants – Volume 1: Bituminous Coal and Natural Gas to Electricity-Revision 2, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2010b. Life Cycle Analysis: Existing Pulverized Coal (EXPC) Power Plant, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2010c. Life Cycle Analysis: Integrated Gasification Combined Cycle (IGCC) Power Plant, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2010d. Life Cycle Analysis: Natural Gas Combined Cycle (NGCC) Power Plant, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2010e. Life Cycle Analysis: Supercritical Pulverzied Coal (SCPC) Power Plant, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2010f. Methane Emissions from Mining Illinois Basin Coals, Quality Guidelines for Energy Systems Studies, Pittsburgh, PA.

NETL, National Energy Technology Laboratory, 2014. Life Cycle Analysis of Natural Gas Extraction and Power Generation, Pittsburgh, PA.

O'Donoughue, P.R., Heath, G.A., Dolan, S.L., Vorum, M., 2014. Life cycle greenhouse gas emissions of electricity generated from conventionally produced natural gas. J. Ind. Ecol. 18, 125–144.

O'Sullivan, F., Paltsev, S., 2012. Shale gas production: potential versus actual greenhouse gas emissions. Environ. Res. Lett., 7.

Odeh, N.A., Cockerill, T.T., 2008. Life cycle GHG assessment of fossil fuel power plants with carbon capture and storage. Energy Policy 36, 367–380.

Saghafi, A., 2012. A Tier 3 method to estimate fugitive gas emissions from surface coal mining. Int. J. Coal Geol. 100, 14–25.

Singh, B., Stromman, A.H., Hertwich, E., 2011a. Life cycle assessment of natural gas combined cycle power plant with post-combustion carbon capture, transport and storage. Int. J. Greenh. Gas Control 5, 457–466.

Singh, B., Strømman, A.H., Hertwich, E.G., 2011b. Comparative life cycle environmental assessment of CCS technologies. Int. J. Greenh. Gas Control 5, 911–921.

Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L., 2007. The Physical Science Basis, Climate Change. IPCC, Cambridge University Press, Cambridge, UK.

Stephenson, E., Doukas, A., Shaw, K., 2012. Greenwashing gas: might a 'transition fuel' label legitimize carbon-intensive natural gas development? Energy Policy 46, 452–459.

Su, S., Han, J.Y., Wu, J.Y., Li, H.J., Worrall, R., Guo, H., Sun, X., Liu, W.G., 2011. Fugitive coal mine methane emissions at five mining areas in China. Atmos. Environ. 45, 2220–2232.

Sørstrøm, S.E., 2001. Konsekvensutredning for Svea Nord, SVEA rapporten. Store Norske Spitsbergen Kulkompani.

Tukker, A., Koning, A.D., Wood, R., Hawkins, T., Lutter, S., Acosta, J., Rueda-Cantuche, J.M., Bouwmeester, M., Oosterhaven, J., Drosdowski, T., Kuenen, J., 2013. EXIOPOL – development and illustrative analyses of a detailed global multiregional environmentally-extended supply and use table/input-output table. Econ. Syst. Res., 25.

UNFCCC, United Nations Framework Convention on Climate Change, 2004. Guidelines for the Preparation of National Communication by Parties Included in Annex I to the Convention, Part I: UNFCCC Reporting Guidelines on Annual Inventories (Following Incorporation of the Provision Of Decision 13/CP.9), Buenos Aires.

UNFCCC, United Nations Framework Convention on Climate Change, 2012. National Inventory Submissions 2012. UNFCCC.

United Nations, 1992. United Nations Framework Convention on Climate Change. United Nations, Rio de Janeiro.

Veltman, K., Singh, B., Hertwich, E.G., 2010. Human and environmental impact assessment of postcombustion CO_2 capture focusing on emissions from amine-based scrubbing solvents to air. Environ. Sci. Technol. 44, 1496–1502.

Weber, C.L., Clavin, C., 2012. Life cycle carbon footprint of shale gas: review of evidence and implications. Environ. Sci. Technol. 46, 5688–5695.

Whitaker, M., Heath, G.A., O'Donoughue, P., Vorum, M., 2012. Life cycle greenhouse gas emissions of coal-fired electricity generation: Systematic review and harmonization (vol 16, pg S53, 2012). J. Ind. Ecol. 16, 966.

Zapp, P., Schreiber, A., Marx, J., Haines, M., Hake, J.F., Gale, J., 2012. Overall environmental impacts of CCS technologies – a life cycle approach. Int. J. Greenh. Gas Control 8, 12–21.

Appendix B: Paper II

Bouman, E. A., M. M. Øberg, and E. G. Hertwich. 2015b. Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES). *Under review with Energy*

Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES)

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ABSTRACT

Using Life Cycle Assessment, we discuss the environmental impacts associated with a Compressed Air Energy Storage (CAES) system as a means of balancing the electricity output of an offshore wind farm with a capacity of 400 MW. We model both conventional CAES and adiabatic CAES (ACAES), with target for baseload production of respectively 200 MW and 150 MW. Results for the CAES system show that wind power production and natural gas combustion are main contributors to the assessed life cycle environmental impact categories. Results for the ACAES system show that wind power production and thermal energy storage are significant contributors. Our main finding is that ACAES has a lower environmental impact than CAES due to the lack of combustion of natural gas. If energy storage size and corresponding thermal energy storage capacity for ACAES are increased the difference in environmental impacts between CAES and ACAES becomes smaller. We find that in comparison with impacts from the average European mix, both storage configurations in our base case have low impacts per kWh electricity delivered to the grid, with the exception of metal depletion potential.

1. INTRODUCTION

Increased implementation of renewable energy, such as wind and solar energy, has clear global environmental benefits [1], but causes unpredictability in power generation and reduces regulatory capacity in the power grid. When renewable power penetration, such as photovoltaic and wind power, is significant, energy storage technologies can be used to address grid issues caused by intermittent power generation. For the International Energy Agency 2DS scenario [2], global wind power production is expected to have a share of 11.6% in 2030 and up to 14.8% in 2050. In absolute terms, this corresponds to an annual electricity production of 3599 TWh and 6145 TWh respectively. A number of researchers [3-5] have pointed to utility scale energy storage technology as a means to enhance grid stability.

Examples of energy storage technologies that could provide significant gains for power grid balancing are flywheels, large scale batteries, pumped hydro storage (PHS) and compressed air systems [4]. At present, except for pumped hydro storage, little energy storage on utility scale level is deployed worldwide. However, the development of PHS is limited due to the lack of suitable sites. CAES is large and powerful enough to store energy on utility scale level and is reported to be much less expensive than other storage systems, approximately half of the costs of lead-acid batteries [5].

In conventional CAES, excess wind power is used to compress air and store this underground under high pressures in a suitable geologic formation such as salt domes and aquifers. The compressed air is subsequently heated and released through high and low pressure expanders to generate electricity. Part of the compressed air is used in a natural gas turbine that produces both electricity and heat necessary for heating the compressed air flow before expansion. The pressure of the air decreases the need for input compression of the natural gas turbine, resulting in an increase of the efficiency of natural gas power generation [6].

An alternative compressed air technology that does not require a fossil fuel is adiabatic CAES (ACAES, also referred to as advanced adiabatic CAES). The ACAES system stores the released thermal energy during compression of the air in a separate thermal energy storage system. This heat is subsequently used to reheat the air prior to expansion, making the system independent from the need for a natural gas inflow [7].

To date, two conventional CAES systems are functional and operating. One plant, located in Huntorf, Germany, has been running since 1978 with a capacity of 290 MW and storage capacity of 3 hours. A 110 MW system with a considerable longer storage capacity of 26 hours is located in McIntosh, Alabama [6]. Both CAES plants use solution-mined salt caverns as air storage location. A small scale adiabatic 2 MW, 500 MWh storage plant was commissioned in Gaines, Texas in late 2012 [8]. A demonstration plant operating on the adiabatic CAES principle is scheduled for completion in 2016 [9].

Despite the limited amount of functional operating projects, compressed air energy storage has been discussed extensively in the scientific literature. In a range of energy storage option reviews, CAES was identified as a low-cost option for utility scale energy storage, with suitable power rating, storage capacity and duration, low self-discharge, high efficiency, and a mature level of technology [3-5, 10-12]. Others focused on modeling the competition between CAES systems and conventional fossil fuel baseload power plants. It is shown that the competitiveness of wind power with CAES systems is highly dependent on effective fossil fuel costs and GHG emissions costs [13]. In the case of Denmark, it has been shown that CAES plants cannot reduce excess electricity production by itself, but can save investments in power plant capacities in the system. In order to be economically feasible plants must be operated on both the spot market and regulating power market [14] with earnings on the spot market likely to be in the range of 80-90 % of optimum since the fluctuations in market prices are not known [15]. Though the intermittent character of wind power presents a challenge to utilization of wind power on an industrial scale, the CAES case studies indicate that the cost of balancing intermittency with CAES is affordable [16].

The potential environmental benefits of implementing CAES systems are discussed Refs. [17] and [18]. Life Cycle Assessment (LCA) methods were employed to study the energy requirements and greenhouse gas (GHG) emissions for PHS, CAES and battery storage. Total GHG emissions, excluding primary electricity generation, were reported to be 292 g CO_2-eq/kWh [17], with most of the direct emissions resulting from the combustion and transportation of natural gas to the CAES site. However, power from the CAES system is only employed when generated wind power is not sufficient to meet demand. When evaluated on a constant production basis, the combination of directly used wind power and additional CAES power during low wind production decreases the life cycle GHG emissions to a range of 66 - 104 g CO_2-eq/kWh, depending on the total system operating capacity factor [18]. The GHG impacts of offshore wind in combination with adiabatic CAES substituting fossil based power generation in Germany are discussed in Ref. [19] using a consequential environmental systems analysis, but the contribution of ACAES is almost negligible as it is assumed that only 1% of the total offshore wind energy is led trough the ACAES storage cycle.

The aim of this article is to quantify the (potential) life cycle environmental impacts associated with two types of CAES systems, conventional CAES and adiabatic CAES, which are coupled to an offshore wind power plant and provide baseload power to the grid. These systems are assessed with three different types of air storage reservoirs: a large hard rock mined cavern, a salt mine dome, and aquifer storage. These are considered to be the best options for CAES, as the geostatic pressure facilitates the high pressures needed for the containment of the air [10]. Other options for storage could be underground piping, but this is considered to be an unlikely option for the required storage volume needed. Preliminary results of this article were presented at a conference [20]. Here, however, a fully revised Life Cycle Inventory is presented, and an energy storage model in order to improve the scaling of the (A)CAES system with respect to lifetime energy production is included. The (A)CAES system size is scaled to balance the output of a wind power facility that has a total capacity of 400 MW. In the following section we describe the methods and life cycle inventory of the compressed air energy storage systems. Subsequently, we present and discuss the results of the life cycle impact assessment.

2. METHODS

A process-based life cycle assessment (LCA) model was employed to model the potential environmental impacts of several compressed air energy storage systems. Ecoinvent v2.2 is used as a background database [21]. The results from the Life Cycle Inventory were characterized using the ReCiPe v1.08 hierarchist impact assessment method [22]. In our model, the fugitive emissions for the fossil fuel extraction processes were updated with emissions data published by Burnham [23] to obtain a better representation of associated impacts of fossil fuel extraction. Calculations were made with Matlab in combination with a Microsoft Excel interface for data input. In this section, we present the key plant characteristics and describe the material and transportation requirements of the different storage systems. The full inventory data are available in the supplementary material, section S2 and more detailed information on the LCA model set-up can be found in Ref. [1].

2.1 Key plant characteristics

We model a wind power facility with a power rating of 400 MW. We simulate the output of this facility based on measured output generation data of Belgium wind power plants in 2013 that were linearly scaled-up to match the power rating of 400 MW [24]. Data were obtained for both onshore and offshore wind power facilities. The annual generated wind power for a 400 MW onshore facility was found to be 822 GWh, corresponding to a capacity factor of 23%. Annual production numbers for an offshore facility of the same power rating are found to be significantly higher at 1424 GWh and a capacity factor of 40%. Therefore, for our base case, we model a wind power plant on an offshore location, with each wind turbine having a rating of 5 MW. The wind turbines are modeled based on an inventory presented in Refs. [1, 25] that was adapted to match the 40% capacity factor (from an original capacity factor of 37.5%). Each turbine has a concrete gravity based foundation and a lifetime of 25 years. Supply of spare parts and replacement are included in the inventory. The connection to the grid and CAES system is provided by 50 km submarine, 10 km underground, and 10 km overhead cabling [1, 25].

The purpose of the energy storage system is to stabilize the intermittency of the wind power, such that the *wind + (A)CAES* system can act as a baseload plant with a capacity factor that is equal to 80% or higher. The CAES system described in this work is based on a design by the Energy Storage Power Corporation [26] and has a typical electricity demand for air compression of 0.67 kWh per kWh of electricity output. In addition 4.2 MJ of natural gas is burned in the gas turbine during the generation of electricity [13]. The ACAES system is based on a design presented in Ref. [27] and has an electricity consumption of 1.38 kWh per kWh electricity produced. Both the CAES and ACAES system have a different power rating than required by our model and are rescaled to meet the requirements of balancing a 400 MW power plant.

The total annual production of a *wind + (A)CAES* system depends on several factors. For a fixed wind power annual production, as specified by the generation data, the target power rating and total energy storage capacity determine the total system production and final system capacity factor. In order to determine the storage capacity and target power rating for our base system, we use a simple model to calculate the amount of electricity produced from the wind + (A)CAES system based on the following rules:

i) When produced wind power is above the target power rating, the target power is fed directly to the grid and the surplus is used to compress air and store the energy.

ii) When wind power is below the target, the (A)CAES system compensates for the shortfall.

In addition the energy storage level is calculated, for which two more rules apply:

iii) When energy storage is at maximum capacity (full cavern) and wind power is above target, the surplus of wind power is curtailed.

iv) When the storage is depleted and wind production is below target level, electricity is directly fed to the grid and the (A)CAES shall be on stand-by.

A graphical representation of the model decisions and outcomes is depicted in Figure S1 in the supporting information. By default, our model data starts in January and ends in December. This starting point is of limited influence (>2%) on the calculated capacity factor at storage capacities up until 25 GWh. This is discussed further in Figure S5 of the supplementary material.

Using the above described model we can calculate the system capacity factor as function of storage capacity and target power rating. The initial storage capacity is set to 80%. In Figure 1, the system capacity factor is plotted as function of storage capacity and target output for a 400 MW offshore wind plant with both CAES and ACAES system. It can be seen that when a low target output is required, the system capacity factor approximates unity. However, for higher target outputs, the capacity factor will decrease significantly. Higher capacity factors can also be achieved by increasing storage capacity. The effect on capacity factor of increased capacity is relatively large for low storage capacities, but tends to flatten out when the capacity exceeds the equivalent of two days maximum wind power production, i.e. 19.2 GWh. Quick depletion of the energy storage at high output targets significantly decreases the operating time and effectiveness of the energy storage, and results in a low capacity factor.

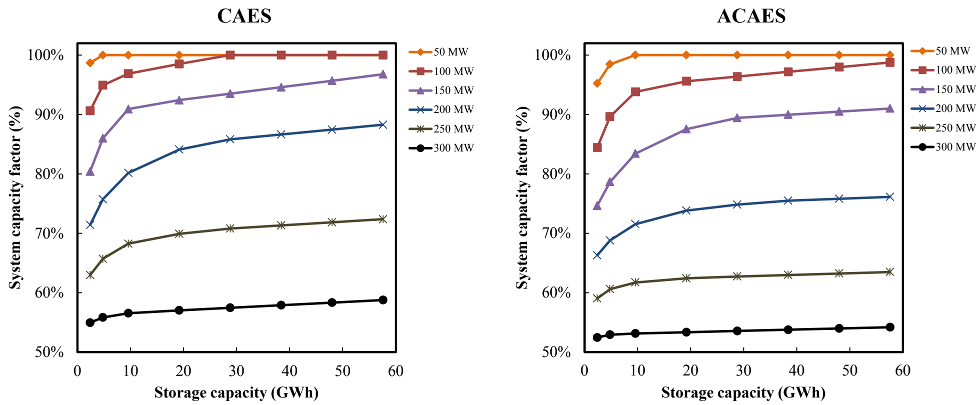

Figure 1: The capacity factor of the (adiabatic) compressed air energy storage in conjunction with wind power production is plotted as a function of storage capacity and target power rating. A low target output results in a high capacity factor. The effect of increasing storage capacity on the capacity factor tends to flatten out when storage capacity exceeds the equivalent of two days wind power production at full capacity.

The opposite effect can be observed for a low target output. Surplus energy will be stored and the energy storage will be at maximum capacity relatively quickly, so that as a result a significant amount of electricity needs to be curtailed. This effect is illustrated in Figure 2, which shows the (intermittent) wind power production and *wind + CAES* system output for a target rating of 100, 150 and 200 MW, and corresponding curtailed electricity for a storage capacity of 19.2 GWh. The annual curtailed electricity as percentage of produced wind power is respectively 45%, 22%, and 5%. The annual system capacity factors are 100%, 95%, and 86%, respectively.

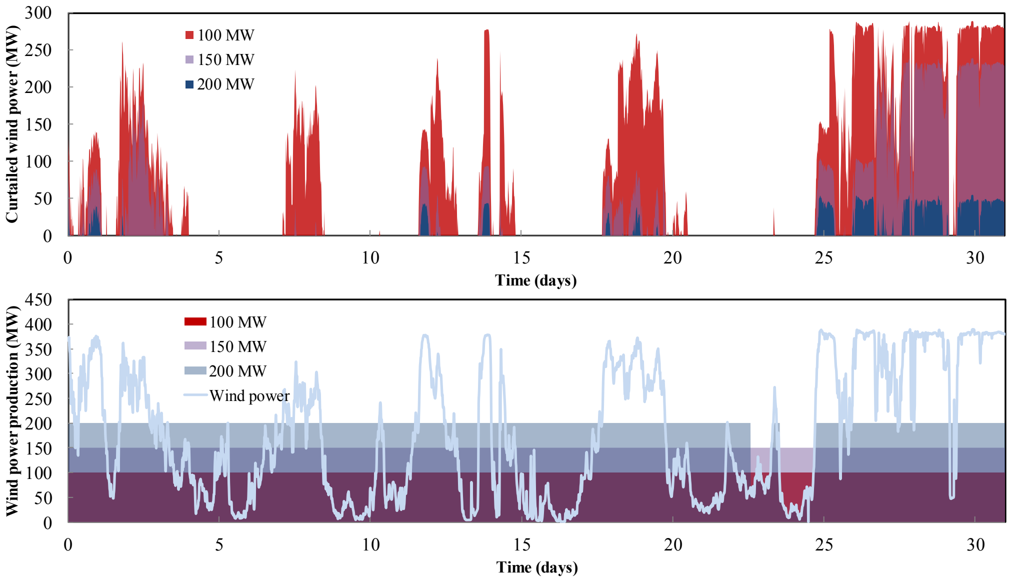

Figure 2: Wind power production, target power output, and curtailed wind power over time for a conventional CAES + wind system with a maximum storage capacity of 19.2 GWh.

Based on our model, we chose to select for our base case a target output of 200 MW for the CAES system and a target output of 150 MW for the ACAES system. The storage capacity is chosen to be equivalent to two days of wind power at full capacity, or 19.2 GWh. The key plant characteristics are summarized in Table 1.

Table 1: Key plant characteristics

	CAES		ACAES	
	Value	Unit	Value	Unit
Wind power input	0.67	kWh/kWh	1.38	kWh/kWh
Natural gas input	4.2	MJ/kWh	-	-
System target rating	200	MW	150	MW
System annual production	1473	GWh	1150	GWh
System capacity factor	84.1	%	87.5	%
Wind power plant capacity	400	MW	400	MW
Wind power plant annual production	1424	GWh	1424	GWh
Curtailed wind power	6.3	%	11.4	%
Air storage pressure	55	bar	138	bar
Hard rock air storage volume	1.49E+06	m^3	5.60E+05	m^3
Salt based cavern air storage volume	6.48E+06	m^3	2.43E+06	m^3

2.2 Air Storage properties

Three different types of underground air storage are considered: A porous rock formation, a mined hard rock storage facility, and a leached out salt dome. The storage volume requirement is related to the operating conditions of the air storage systems. In this article, we discuss a fixed rigid storage volume. Due to the need for pressure compensation, the salt based cavern is approximately 4.3 times larger than a hard rock based cavern [6]. Assuming ideal gas behavior, we calculate the storage volumes of the hard rock cavern and salt dome. These are presented in Table 1. As the reported storage pressure of ACAES is significantly larger than the storage pressure of CAES, the ACAES storage volume is smaller than the CAES storage volume. The porous rock formation does not have to be constructed and is considered to be of sufficient volume. The air storage is modeled with a lifetime of 100 year.

The construction of the air well is the same for all three types of storage and modeled after an exploration and production as available in ecoinvent. The well has a length of 680 meter to reach the storage depth. For air storage in porous rock, it is assumed that the required volume is covered by existing cavities underground and no other items are inventoried. The limestone mining process in ecoinvent was adapted to model mining of hard rock. The total mass of the storage volume was determined using a density of 2500 kg/m^3 in order to calculate the mining requirements. Diesel and light fuel oil are used in the mining process at 18 kJ/kg and 0.59 kJ/kg rock mined. In addition dynamite for blasting is included at 0.073 g/kg rock mined [28].

In order to produce a salt dome, a well has to be drilled to the starting depth of the salt dome and the storage volume has to be leached out. The salt dome leaching process incorporates the energy use for pumping water and brine, the construction of the pump and pipeline infrastructure and the disposal process of the resulting brine. Ocean water is assumed to be used for the leaching as this would be a logical choice for the connection of a compressed air storage facility to an offshore wind facility. The energy requirement related to cavern development from a salt dome is reported to be 16.2 GJ/MWh storage capacity [17] and is supplied by the Norwegian electricity mix. It is assumed that the salt brine is of sufficient quality to allow release in the marine environment without treatment and a 1 km pipeline is included for transport of the salt water to the sea.

2.3 Plant construction, machinery, and equipment

We assume that the same type of building infrastructure is used for both the conventional and adiabatic CAES. We rescale data from Ref. [29] and Ref. [30] for natural gas power plants and CAES storage. The plant construction inventory is presented in the supplementary material. Material requirements are taken from Ref. [30], and energy requirements for construction from ecoinvent. Both the CAES and ACAES system use a compressor and high and low pressure expanders. For the purpose of modeling we assume the material requirements for this equipment to be similar to those of a steam turbine. The compressor and expanders are modeled using a combination of iron, steel, copper and rock wool [29, 31].

2.3.1 Gas turbine

The CAES system makes use of a gas turbine for heating up the outflow of air from the storage location. The gas turbine model is a downscaled version (to 80 MW) of a 265 MW Siemens turbine as described in Ref. [31]. The inventory consists of iron and steel, copper, ceramics, plastics, and organic substances.

2.3.2 Thermal Energy Storage

In ACAES system the preheating of the airflow before expansion is done by making use of thermal energy storage (TES). The TES consists of two separate thermal cycles, with each cycle being connected to a compressor and an expander. The heat generated during air compression is stored in a thermal mass with a specific heat of 3.33 kJ/(kg.K) consisting of a thermal mineral oil. Storage tanks in the system act as buffer for the thermal mass. The material requirement for the thermal energy storage systems is based on an analysis for thermal energy storage for a solar power plant presented by Nandi et al. [32] The size of the storage tanks and necessary amount of thermal mass is determined by the target system requirements for storage set above, i.e. 19.2 GWh. The inventory for the thermal energy storage is presented in the supplementary material.

All materials in the inventory are transported by train from a location in Europe over a distance of 1260 km, which corresponds roughly to the air distance between central Germany and Trondheim. An additional 140 km is added for lorry transportation.

3. RESULTS AND DISCUSSION

3.1 Life cycle impact assessment

In this section, we present the environmental impacts of (A)CAES for selected impact categories on a *per kWh* basis. A foreground contribution analysis, a breakdown of the life cycle impacts into the relative contribution for each of the modeled components and processes, for the CAES and ACAES (both using porous rock storage) is shown in Figure 3. For the CAES system the main contributors for most of the impact categories are the actual wind power generation and the natural gas combustion, which account for more than 95%. Plant construction contributes in the order of 1-4%, for most impact categories except for land occupation to which construction contributes 22%. Impacts related to machinery and equipment, and components transport are negligible. For the ACAES system the main contributors are wind power generation and thermal energy storage, followed by plant construction. The thermal energy storage has a relatively large share of the impact due to its high material requirements compared to the other components. The thermal mass required to satisfy the operating constraints of the ACAES system also has consequences for ancillary materials, such as concrete, necessary to construct the tanks and foundation for its storage.

Total impacts of 1 kWh electricity generation from an (A)CAES system are also specified in Figure 3. GHG emissions, particulate matter formation, photochemical oxidant formation, and terrestrial acidification are significantly lower for the ACAES system than for the CAES system. However, the ACAES system requires more wind power per kWh output and has a lower target power rating than the CAES system, which results in higher material requirements per unit output. As a consequence, impacts that are not heavily influenced by fossil fuel combustion increase significantly and potentially double.

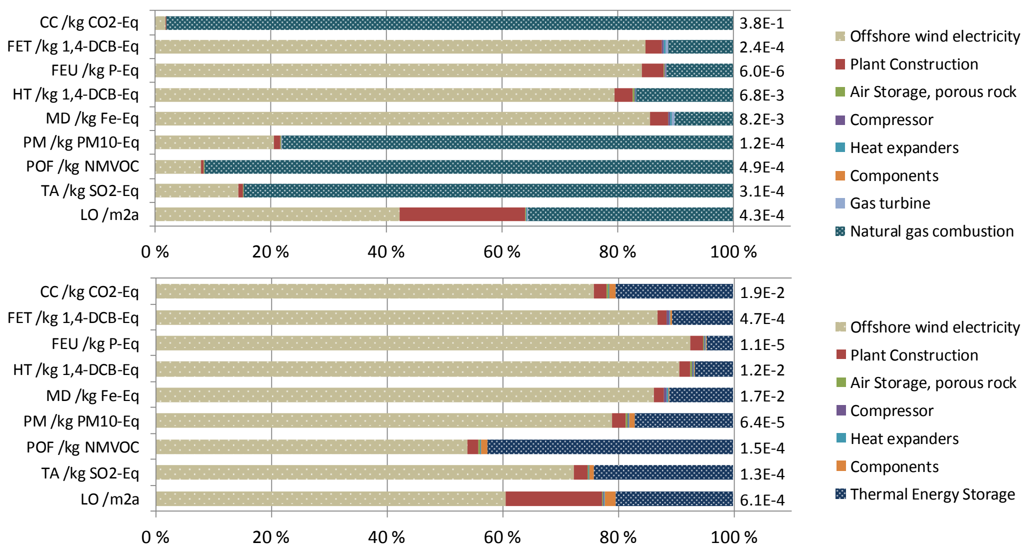

Figure 3: Contribution analysis for 1 kWh electricity generation provided by a CAES (top) and ACAES (bottom) system connected to an offshore wind power plant. Impact categories: CC - climate change, FET - freshwater ecotoxicity potential, FEU - freshwater eutrophication potential, HT - human toxicity potential, MD - metal depletion potential, PM - particulate matter formation potential, POF - photochemical oxidant formation potential, TA - terrestrial acidification potential, LO - agricultural and urban land occupation potential.

We do not show here the additional impact of mined hard rock volume or leached out salt dome. These results are presented in Figures S3 and S4 in the supplementary material. The additional impact associated with the different storage types is practically negligible. An increase between 1% and 2% can be seen for particulate matter, terrestrial acidification, photochemical oxidant formation, and land occupation. Even though energy requirements for constructing underground volumes are significant, the energy stored and discharged over the lifetime of 100 years is much larger. This is illustrated in Figure S2 of the supporting information were the ratio between energy for construction of the cavern is plotted against the lifetime assumption for the cavern.

3.2 Impacts of baseload wind power

Not all electricity produced by the wind park is stored in order to reach the required target output. Of the annual production of the *wind + CAES* system, 29% of electricity comes from the storage and 71% is wind power fed directly to the grid at target levels. For the *wind + ACAES* system, 25% of electricity is stored and 75 % of wind power is fed directly to the grid. The impacts associated with baseload electricity generation from a *wind + (A)CAES* system are therefore lower than the impact associated when assessing only the storage route. In order to show the results of wind power and energy storage in perspective, we compare in this section the impacts from *wind + (A)CAES* to impacts from the average European mix and to impacts related to electricity production from a natural gas combined cycle (NGCC) power plant with carbon dioxide capture and storage.

In Figure 4, the impacts of offshore wind, wind in combination with (A)CAES, and electricity from a NGCC plant with carbon dioxide capture and storage (CCS) technology, are plotted relative to impacts associated with the European electricity mix. Results for both the NGCC plant and European mix were obtained from Ref. [1]. The absolute results for the impact assessment are shown in Table S6 of the supplementary material. From Figure 4 and Table S6 we can observe that balancing wind power with a compressed air storage system in order to supply baseload power will have a minor increase of impacts when compared to impacts of wind power alone. An exception occurs for the *wind + CAES* system as the following impact categories show a minor decrease compared to wind power: freshwater eutrophication potential, eco- and human toxicity potential, and metal depletion potential. These are slightly reduced as a result of the combustion of natural gas. When *wind + CAES* is compared to the NGCC with CCS, impacts are slightly lower, with the exception of metal depletion, land occupation and eutrophication. Though adding CCS technology to a natural gas power plant increases its non GHG related impacts, these three impacts are still higher for the *wind + (A)CAES* system.

The results presented in Figure 4 point to one of the challenges related to comparative life cycle impact assessment. We compare what are considered to be low-carbon technologies against average impacts of the current electricity mix. For wind systems as well as NGCC with CCS it can be seen that the GHG emissions are lower, and that these technologies in fact enable a reduction in impact for at least most impact categories presented. At the same time, we cannot identify one clear winner; each of the technologies comes with environmental trade-offs.

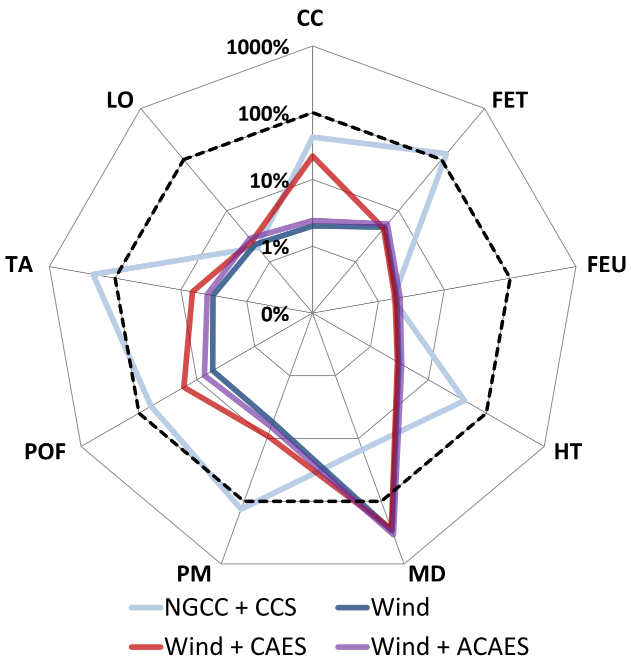

Figure 4: Impacts for wind, *wind + CAES*, *wind + ACAES*, and NGCC + CCS, relative to impacts from the European mix. Impact categories: CC - climate change, FET - freshwater ecotoxicity potential, FEU - freshwater eutrophication potential, HT - human toxicity potential, MD - metal depletion potential, PM - particulate matter formation potential, POF - photochemical oxidant formation potential, TA - terrestrial acidification potential, LO - agricultural and urban land occupation potential.

The GHG results reported in this article for the *wind + CAES* system (118 g CO_2-eq/kWh), are in the same order of magnitude as reported by Denholm *et al*. (104 g CO_2-eq/kWh).[18] However, the impacts of the natural gas delivery and combustion are significantly higher for the CAES system presented here (379 g CO_2-eq/kWh electricity from CAES vs. 285 g CO_2-eq/kWh[18]) as a result of updating GHG emissions in the natural gas production chain (for method see Ref. [1]).

3.3 Influence of target rating and energy storage capacity

In order to investigate the effect of the target rating and the energy storage capacity of the *wind + (A)CAES* system on the capacity factor and total system production, we performed an optimization routine on the wind power model, using Matlabs internal constrained nonlinear optimization function *fmincon*, which makes use of an interior-point algorithm. The purpose of optimization was to maximize annual production (or rather to minimize the square of the difference between annual output and theoretical maximum output) while keeping the capacity factor for the overall *wind + (A)CAES* system above 80 %. As is previously observed, the overall capacity factor, and therefore overall production, increases with increasing storage capacity, but not with target rating. Optimizing for maximum annual production leads the model to select the highest energy storage capacity possible in order to create the largest buffer, while maximizing target output rating. As the capacity factor is used here as a constraint, the result of this optimization is that it is pushed to its minimum allowed value, i.e. 80%. Higher target outputs would result in higher system production, but cause significant downtime due to depletion of the storage capacity, thus opposing the balancing principle of the CAES system. This is illustrated in Figure S6 of the supplementary material. We plot the annual electricity production of both CAES and ACAES systems for various storage capacities as function of target power rating. Additionally, the 80% capacity factor threshold is given. It can be seen that high annual production can be reached at high storage values, but that the rate at which production increases drops tremendously for higher storage capacities. Given the 80 % capacity factor threshold, target power ratings could have been 216 and 177 MW instead of 200 and 150 MW for our model. Alternatively, the storage capacity could have been decreased. The influence of decreasing, or increasing, the energy storage capacity is limited for the CAES system, but significant for the adiabatic storage configuration. A contribution analysis for a doubling of energy storage size to 38.4 GWh is shown in Figure S7 of the supplementary material. A comparison between impact totals is shown in Tables S7 and S8 of the supplementary material. As a large part of the impacts for ACAES is related to the material intensive thermal energy storage system, which in turn scales with the air storage capacity, there is a significant trade-off between environmental impacts per unit production and the technological possibility to maximize wind baseload power generation. The contribution analyses presented in this article show that for CAES the infrastructural requirements related to storage are not significant as most impacts result from wind power production and natural gas combustion. Air storage capacity could be maximized in order to divide wind power impacts over as much electricity production as possible. For ACAES, the increase of storage capacity is only preferable when total annual production increases at a quicker rate, so that thermal energy storage related impacts can be divided over a larger production.

4. CONCLUSIONS AND OUTLOOK

In this article, we calculate life cycle environmental impacts related to balancing offshore wind power with a CAES and an ACAES system. We find that both compressed air systems have a significant reduction in impacts when compared to impacts of the average electricity mix, with the exception of impacts that are influenced heavily by the infrastructural intensity of wind power. Adiabatic CAES overall has better environmental performance than CAES, but this is heavily influenced by the thermal energy storage size, which has to be balanced with the target output rating of the storage system. In the base case described in this article, a relatively large storage capacity of 19.2 MWh is used. In wind-storage configurations where intermittency balancing can be provided by smaller storage capacities the environmental advantages of the ACAES system over the CAES system become more pronounced as an equally smaller thermal storage is required.

It is unsure what role compressed air technology will play in securing baseload renewable power generation in future electricity generation systems. The capital investment required, in combination with finding a suitable geological location, might prove to be a significant impediment for large scale implementation. The results presented in this article show that compared to the average electricity mix, or even natural gas technology with carbon dioxide capture technology, compressed air storage in combination with wind can significantly reduce impacts for a number of impact categories while maintaining baseload properties. More generally, the results point to the differences that can be observed between infrastructure intensive and fuel intensive electricity generation. Impact categories that are influenced by infrastructure related processes (e.g. resource extraction, material production), such as land occupation, eutrophication, and metal depletion have higher results for wind systems than for natural gas power. Impact categories influenced by fuel combustion and associated emissions, such as climate change, particulate matter, and photochemical oxidant formation show lower results for the wind systems than for natural gas power. A similar effect can be observed for the differences between impacts of the ACAES and CAES systems, with the ACAES system being more material intensive than the CAES system.

If greenhouse gases, toxicity impacts, and particulate matter emissions are of particular concern, impacts are in general lower for the material intensive wind power technologies than for (fossil) fuel intensive power technologies. One could expect that adding CCS technology to the natural gas turbine in the CAES system would effectively reduce the GHG emissions levels to levels comparable to ACAES, while moderately increasing other impact categories. However, the fugitive GHG emissions associated with the natural gas fuel production chain are likely to remain an issue. The increased metal depletion potential might be reduced by effective application of recycling and reuse of materials, as well as lifetime enhancement.

The different impact profiles of the evaluated technologies make it likely that a combination of technologies has a better overall performance when the mix is evaluated. The challenge is to balance and optimize impacts related to infrastructure with impacts related to (fossil) fuel combustion. Such an analysis with a focus beyond GHG emissions and climate change mitigation would provide insights in the possible development of an electricity generation system with low(est) environmental impacts.

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REFERENCES

[1] Hertwich EG, Gibon T, Bouman EA, Arvesen A, Suh S, Heath GA, et al. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. Proceedings of the National Academy of Sciences. 2014;10.1073/pnas.1312753111.

[2] International Energy Agency. Energy Technology Perspectives 2012: Pathways to a Clean Energy System. 2012.

[3] Beaudin M, Zareipour H, Schellenberglabe A, Rosehart W. Energy storage for mitigating the variability of renewable electricity sources: An updated review. Energy for Sustainable Development. 2010;14(4):302-14.

[4] Chen H, Cong TN, Yang W, Tan C, Li Y, Ding Y. Progress in electrical energy storage system: A critical review. Progress in Natural Science. 2009;19(3):291-312.

[5] Evans A, Strezov V, Evans TJ. Assessment of utility energy storage options for increased renewable energy penetration. Renewable and Sustainable Energy Reviews. 2012;16(6):4141-7.

[6] Succar SW, Robert H. Compressed Air Energy Storage: Theory, Resources, And Applications For Wind Power. 2008.

[7] Jubeh NM, Najjar YSH. Green solution for power generation by adoption of adiabatic CAES system. Applied Thermal Engineering. 2012;44:85-9.

[8] General Compression. Texas Dispatchable Wind Project. 21 March 2014: http://www.generalcompression.com/index.php/tdw1

[9] RWE Power AG. ADELE - Adiabatic compressed-air energy storage for electricity supply. Cologne, Germany2010.

[10] Ibrahim H, Ilinca A, Perron J. Energy storage systems - Characteristics and comparisons. Renew Sust Energ Rev. 2008;12(5):1221-50.

[11] Diaz-Gonzalez F, Sumper A, Gomis-Bellmunt O, Villafafila-Robles R. A review of energy storage technologies for wind power applications. Renew Sust Energ Rev. 2012;16(4):2154-71.

[12] Sundararagavan S, Baker E. Evaluating energy storage technologies for wind power integration. Solar Energy. 2012;86(9):2707-17.

[13] Greenblatt JB, Succar S, Denkenberger DC, Williams RH, Socolow RH. Baseload wind energy: modeling the competition between gas turbines and compressed air energy storage for supplemental generation. Energ Policy. 2007;35(3):1474-92.

[14] Lund H, Salgi G. The role of compressed air energy storage (CAES) in future sustainable energy systems. Energy Conversion and Management. 2009;50(5):1172-9.

[15] Lund H, Salgi G, Elmegaard B, Andersen AN. Optimal operation strategies of compressed air energy storage (CAES) on electricity spot markets with fluctuating prices. Applied Thermal Engineering. 2009;29(5-6):799-806.

[16] Cavallo A. Controllable and affordable utility-scale electricity from intermittent wind resources and compressed air energy storage (CAES). Energy. 2007;32(2):120-7.

[17] Denholm P, Kulcinski GL. Life cycle energy requirements and greenhouse gas emissions from large scale energy storage systems. Energy Conversion and Management. 2004;45(13-14):2153-72.

[18] Denholm PK, Gerald L; Holloway, Tracey. Emissions and Energy Efficiency Assessment of Baseload WInd Energy Systems. Environ Sci Technol. 2005;39:1903-11.

[19] Pehnt M, Oeser M, Swider DJ. Consequential environmental system analysis of expected offshore wind electricity production in Germany. Energy. 2008;33(5):747-59.

[20] Bouman EA, Øberg MM, Hertwich EG. Life Cycle Assessment of Compressed Air Energy Storage. The 6th International Conference on Life Cycle Management in Gothenburg 20132013.

[21] Dones R, Bauer C, Bolliger R, Burger B, Faist Emmenegger M, Frischknecht R, et al. Life Cycle Inventories of Energy Systems: Results for Current Systems in Switzerland and other UCTE Countries. Dübendorf, CH: Swiss Centre for Life Cycle Inventories; 2007.

[22] Goedkoop M, Heijungs R, Huijbregts MAJ, de Schryver A, Struijs J, Van Zelm R. ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. 2013.

[23] Burnham A, Han J, Clark CE, Wang M, Dunn JB, Palou-Rivera I. Life-Cycle Greenhouse Gas Emissions of Shale Gas, Natural Gas, Coal, and Petroleum. Environ Sci Technol. 2012;46(2):619-27.

[24] ELIA. Historical wind power generation data. 27 June 2014: http://www.elia.be/en/grid-data/power-generation/wind-power

[25] Arvesen A, Birkeland C, Hertwich EG. The importance of ships and spare parts in LCAs of offshore power. Environ Sci Technol. 2013;47(6):2948-56.

[26] Nakhamkin M. 150-300-400 MW CAES Plants Based on Various Combustion Turbines. In: Energy Storage & Power Corporation (ESPC), editor. http://www.espcinc.com/library/150-300-400_MW_CAES_Brochure.pdf2008.

[27] Biasi Vd. Fundamental Analyses to Optimize Adiabatic CAES Plant Efficiencies. Gas Turbine World. 2009:26-8.

[28] Kellenberger D, Althaus HJ, Jungbluth N, Künniger T, Lehmann M, Thalmann P. Life Cycle Inventories of Building Products. Final report ecoinvent Data v2.0 No. 7. Dübendorf: Swiss Centre for Life Cycle Inventories; 2007.

[29] Briem S, Blesl M, Fahl U, Ohl M. Fossil gefeuerte Kraftwerke in: Lebenszyklusanalysen ausgewählter zukünftiger Stromerzeugungstechniken. Institut für Energiewirtschaft und Rationelle Energieanwendung, Universität Stuttgart; 2004.

[30] Oeser M. Systemanalyse der Umweltwirkungen hoher Windstromanteile: Heidelberg 2006.

[31] Parthey F. Lebenszyklusanalyse und Bestimmung von Einflussfaktoren zur nachhaltigen Produkt- gestaltung von GuD-Kraftwerken. Brandenburgische Technische Universität Cottbus; 2011.

[32] Nandi BRS, B.; Banerjee, R. Analysis of High Temperature Thermal Energy Storage for Solar Power Plant. IEEE ICESET 2012. Nepal2012.

Supplementary material for the article

Environmental impacts of balancing offshore wind power with Compressed Air Energy Storage (CAES)

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Content

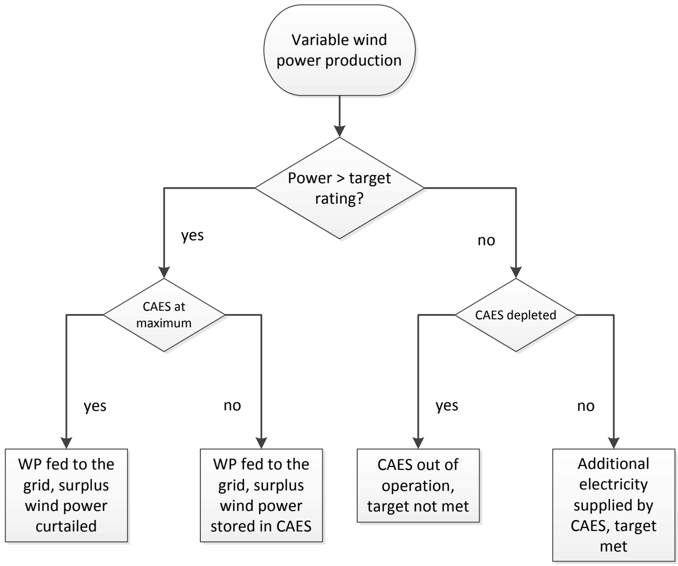


Figure S1: Simplified representation of the storage model structure

S2) Scaling assumptions and Life Cycle Inventory tables

Table S1: Scaling factors

For all inventories it was assumed that the power rating and weight scale linear. The following table summarizes the scaling factors used.

	CAES	unit	ACAES	unit	Reference	unit	Scaling factor CAES	Scaling factor ACAES	Reference
Plant rating	200	MW	150	MW	250	MW	0.80	0.60	Oeser (1)
Compressor cumulative power	134	MW	207	MW	424	MW	0.32	0.49	Parthey (2)
					284	MW	0.47	0.73	Briem (3)
Heat expander cumulative power	124	MW	150	MW	424	MW	0.29	0.35	Parthey (2)
					284	MW	0.44	0.53	Briem (3)
Gas turbine	80	MW	-	-	265	MW	0.30	-	Parthey (2)
Thermal energy storage	-	-	4.8E+07	kg	2.56E+07	kg	-	1.88	Nandi (4)

Table S2: Plant construction inventory

The plant construction inventory is rescaled from data by Oeser.[1]

Plant construction	CAES	unit	ACAES	unit
Unalloyed steel	9.28E+05	kg	6.96E+05	kg
Low alloyed steel	3.95E+05	kg	2.96E+05	kg
Aluminium	1.12E+04	kg	8.38E+03	kg
Copper	5.38E+04	kg	4.04E+04	kg
Concrete	3.79E+06	kg	2.84E+06	kg
Plastics	1.61E+04	kg	1.21E+04	kg
Rock wool	3.19E+04	kg	2.39E+04	kg

Table S3: Compressors and heat expanders

We assume that the material requirements for both compressor and heat expanders are similar to those of a steam turbine. All iron related requirements are based on Parthey[2], copper and rock wool requirements are based on Briem et al.[3]

	CAES		ACAES	
Compressor		unit		unit
Unalloyed steel	3.86E+04	kg	5.96E+04	kg
Low alloyed steel	1.10E+03	kg	1.69E+03	kg
High alloyed steel	2.81E+04	kg	4.34E+04	kg
Cast iron	2.73E+04	kg	4.22E+04	kg
Copper	1.08E+03	kg	1.67E+03	kg
Rock wool	4.34E+03	kg	6.70E+03	kg

	CAES		ACAES	
Heat expanders		unit		unit
Unalloyed steel	3.57E+04	kg	4.32E+04	kg
Low alloyed steel	1.01E+03	kg	1.23E+03	kg
High alloyed steel	2.60E+04	kg	3.14E+04	kg
Cast iron	2.53E+04	kg	3.06E+04	kg
Copper	1.00E+03	kg	1.21E+03	kg
Rock wool	4.01E+03	kg	4.85E+03	kg

Table S4: Gas turbine inventory

For the gas turbine inventory of the gas turbine in the compressed air energy storage system we use a Siemens gas turbine with 265 MW power rating as a proxy.[2]

Gas turbine inventory (80 MW)		unit
High alloyed steel	3.87E+04	kg
Low alloyed steel	1.96E+04	kg
Unalloyed steel	1.49E+04	kg
Cast iron	2.08E+04	kg
Copper	4.17E+01	kg
Aluminium	4.17E+01	kg
Ceramics	1.65E+02	kg
Plastics	2.78E+01	kg
Organic substances	3.68E+02	kg

Table S5: Thermal energy storage

The material requirement for the thermal energy storage systems is based on an analysis for thermal energy storage for a solar power plants presented by Nandi et al. [4]

Thermal Energy Storage inventory		unit
High alloyed steel, stainless	7.82E+05	kg
Low alloyed steel	2.14E+06	kg
Concrete	7.22E+06	kg
Foam glass	6.21E+04	kg
Refractory brick	4.56E+05	kg
Rock wool	1.94E+05	kg
Limestone	8.84E+04	kg
Thermal mass, molten salt proxy, calcium nitrate, as N	1.74E+06	kg
Thermal mass, molten salt proxy, potassium nitratem as K2O	3.51E+07	kg
Thermal mass, molten salt proxy, potassium nitrate; as N	1.12E+07	kg

S3) Additional Results

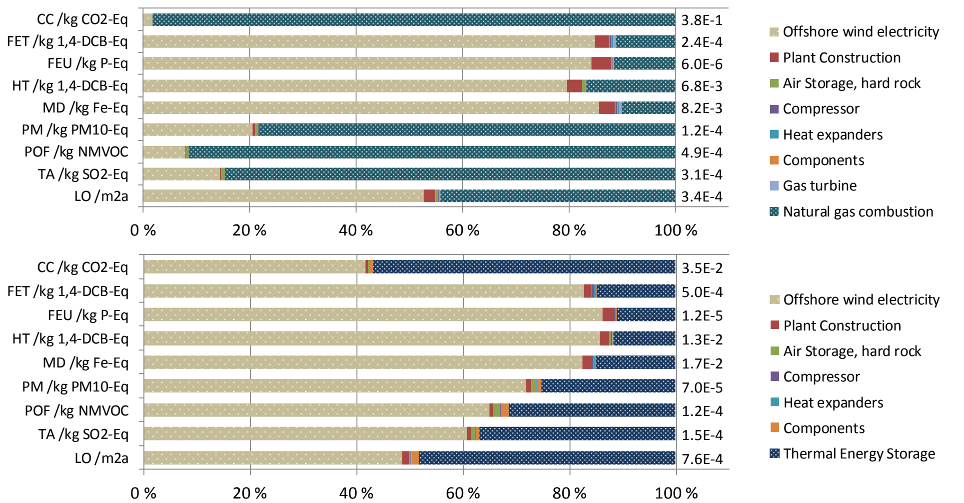


Figure S2: Contribution analysis for 1 kWh electricity generation provided by a CAES (top) and ACAES (bottom) system connected to an offshore wind power plant with a hard rock mined storage volume.

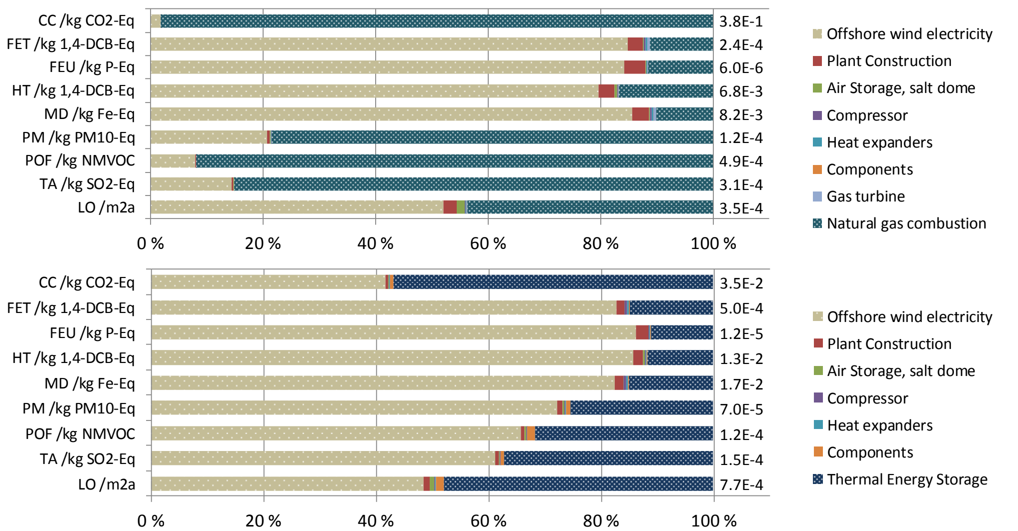


Figure S3: Contribution analysis for 1 kWh electricity generation provided by a CAES (top) and ACAES (bottom) system connected to an offshore wind power plant with a salt dome storage volume.

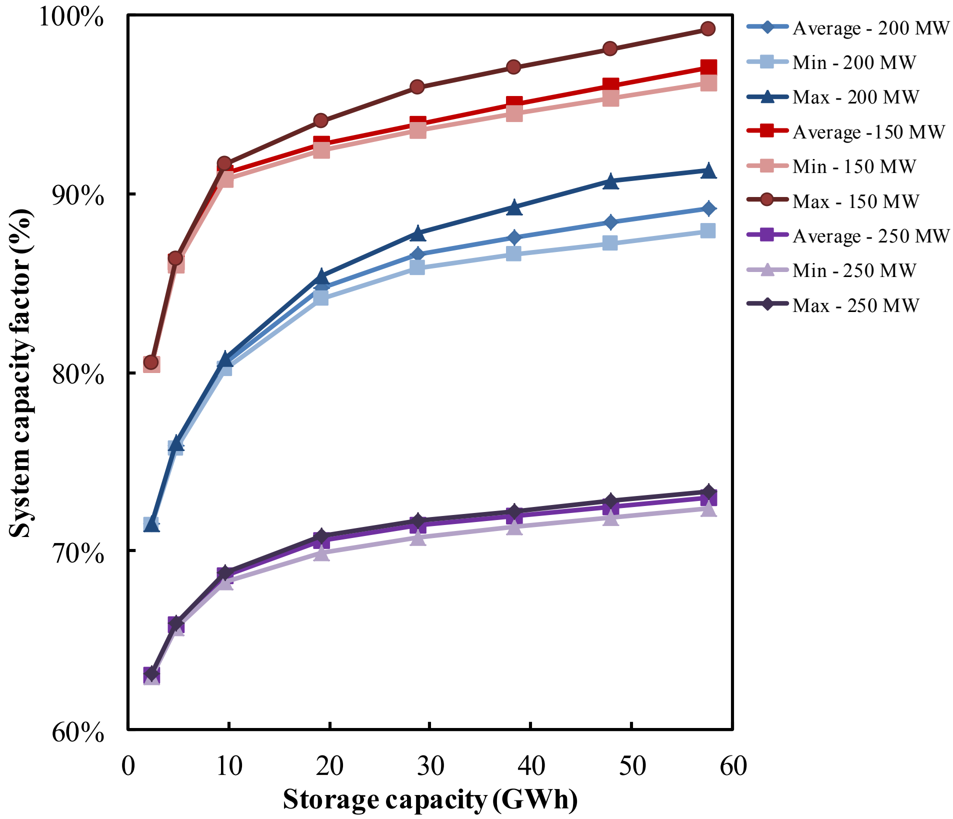
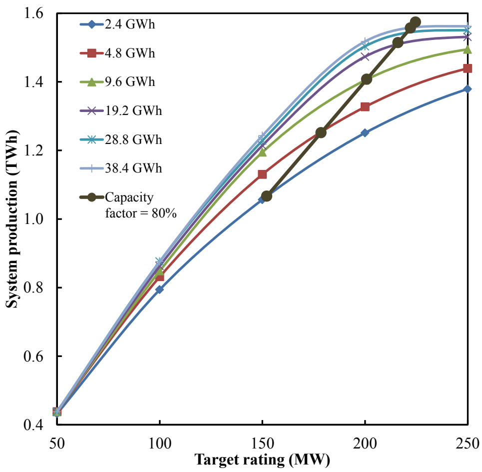


Figure S4: Sensitivity analysis - Influence of the starting point in the wind power time series on the system capacity factor for offshore wind power + CAES storage at three different target ratings.

The data for the offshore wind power plant are available for the year 2013. By default, our model starts in January and ends in December. We have run the model multiple times using different starting points, i.e. March 2013, May 2013, July 2013, September 2013 and November 2013, and calculated the average system capacity factor as function of storage capacity and target rating. In this graph the obtained average is plotted together with the minimum and maximum values for three different target ratings, 150, 200, and 250 MW. For lower storage capacities we see that the variation is relatively small (>2%), but for higher storage capacities the starting point of the time series appears to become more relevant. The system capacity factor is of importance as it influences the annual production.



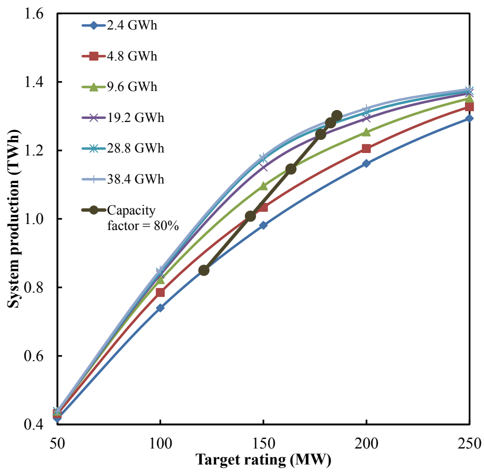
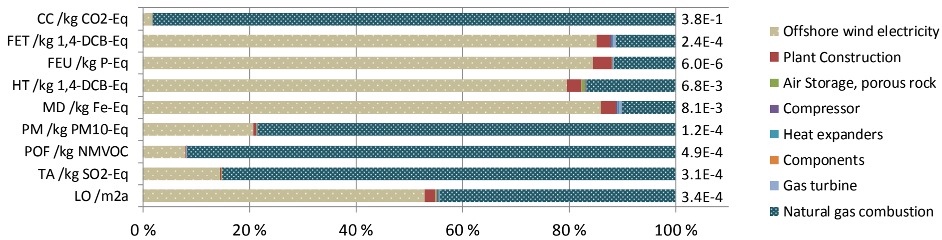


Figure S5: Annual production of a wind power + (A)CAES system as function of the target power output rating and energy storage capacity. Also indicated is also the maximum output at which the system is able to achieve a capacity factor of 80%.



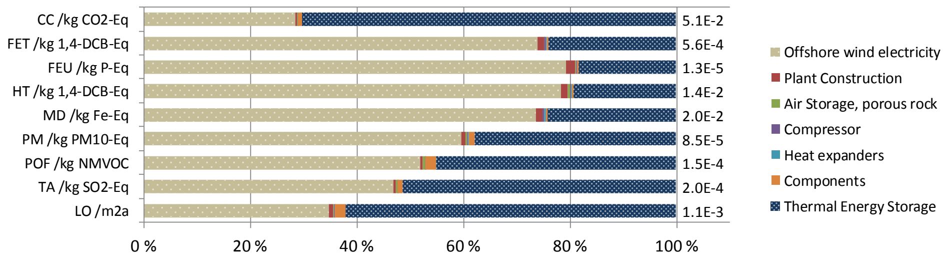


Figure S6: Contribution analysis for 1 kWh electricity generation provided by a CAES (top) and ACAES (bottom) system connected to an offshore wind power plant with a porous rock storage, with a capacity of 38.4 GWh; twice the default size used elsewhere in this paper. Target power ratings are 200 MW for CAES and 150 MW for ACAES.

Table S6: Life cycle environmental impact results for different electricity producing systems

Impact category	**Wind**	**Wind + CAES**	**Wind + ACAES**	**CAES**	**ACAES**	**NGCC + CCS [5]**	**NGCC [5]**
CC /kg CO_2-Eq	1.06E-02	1.17E-01	1.67E-02	3.79E-01	3.50E-02	2.27E-01	5.16E-01
FET /kg 1,4-DCB-Eq	2.99E-04	2.81E-04	3.49E-04	2.36E-04	4.98E-04	8.13E-03	6.33E-03
FEU /kg P-Eq	7.58E-06	7.13E-06	8.71E-06	6.03E-06	1.21E-05	6.63E-06	4.54E-06
HT /kg 1,4-DCB-Eq	8.10E-03	7.72E-03	9.32E-03	6.80E-03	1.30E-02	1.11E-01	8.72E-02
MD /kg Fe-Eq	1.04E-02	9.77E-03	1.22E-02	8.16E-03	1.75E-02	5.28E-04	2.65E-04
PM /kg PM10-Eq	3.66E-05	6.03E-05	4.50E-05	1.18E-04	6.99E-05	8.90E-04	7.38E-04
POF /kg NMVOC	5.80E-05	1.82E-04	7.39E-05	4.85E-04	1.22E-04	6.74E-04	5.62E-04
TA /kg SO_2-Eq	6.77E-05	1.39E-04	8.89E-05	3.12E-04	1.53E-04	4.50E-03	3.67E-03
LO /m^2a	2.69E-04	2.90E-04	3.92E-04	3.41E-04	7.62E-04	2.35E-04	1.63E-04

Table S7: Absolute impacts and relative change for the (A)CAES system as a result of doubling the energy storage capacity

Impact category	**CAES 19.2 GWh**	**ACAES 19.2 GWh**	**CAES 38.4 GWh**	**% change**	**ACAES 38.4 GWh**	**% change**
CC /kg CO_2-Eq	3.79E-01	3.50E-02	3.79E-01	0.00 %	5.12E-02	46.20 %
FET /kg 1,4-DCB-Eq	2.36E-04	4.98E-04	2.35E-04	-0.32 %	5.58E-04	11.90 %
FEU /kg P-Eq	6.03E-06	1.21E-05	6.01E-06	-0.33 %	1.32E-05	8.81 %
HT /kg 1,4-DCB-Eq	6.80E-03	1.30E-02	6.80E-03	-0.01 %	1.42E-02	9.57 %
MD /kg Fe-Eq	8.16E-03	1.75E-02	8.13E-03	-0.37 %	1.95E-02	12.00 %
PM /kg PM10-Eq	1.18E-04	6.99E-05	1.18E-04	0.00 %	8.46E-05	20.99 %
POF /kg NMVOC	4.85E-04	1.22E-04	4.85E-04	0.03 %	1.54E-04	26.73 %
TA /kg SO_2-Eq	3.12E-04	1.53E-04	3.12E-04	0.01 %	1.99E-04	30.39 %
LO /m^2a	3.41E-04	7.62E-04	3.41E-04	-0.14 %	1.06E-03	39.82 %

Table S8: Absolute impacts and relative changes for the *(A)CAES + wind* system as a result of doubling the energy storage capacity.

Impact category	**CAES + wind 38.4 GWh**	**% change**	**ACAES + wind 38.4 GWh**	**% change**
CC /kg CO_2-Eq	1.25E-01	6.28 %	2.15E-02	29.09 %
FET /kg 1,4-DCB-Eq	2.79E-04	-0.53 %	3.69E-04	5.73 %
FEU /kg P-Eq	7.09E-06	-0.52 %	9.09E-06	4.35 %
HT /kg 1,4-DCB-Eq	7.69E-03	-0.34 %	9.76E-03	4.66 %
MD /kg Fe-Eq	9.72E-03	-0.56 %	1.29E-02	5.79 %
PM /kg PM10-Eq	6.20E-05	2.71 %	4.96E-05	10.30 %
POF /kg NMVOC	1.91E-04	4.72 %	8.40E-05	13.60 %
TA /kg SO_2-Eq	1.43E-04	3.53 %	1.03E-04	15.98 %
LO /m^2a	2.91E-04	0.45 %	4.84E-04	23.38 %

S4) References

[1] Oeser M. Systemanalyse der Umweltwirkungen hoher Windstromanteile: Heidelberg 2006.
[2] Parthey F. Lebenszyklusanalyse und Bestimmung von Einflussfaktoren zur nachhaltigen Produkt-gestaltung von GuD-Kraftwerken. Brandenburgische Technische Universität Cottbus; 2011.
[3] Briem S, Blesl M, Fahl U, Ohl M. Fossil gefeuerte Kraftwerke in: Lebenszyklusanalysen ausgewählter zukünftiger Stromerzeugungstechniken. Institut für Energiewirtschaft und Rationelle Energieanwendung, Universität Stuttgart; 2004.
[4] Nandi BRS, B.; Banerjee, R. Analysis of High Temperature Thermal Energy Storage for Solar Power Plant. IEEE ICESET 2012. Nepal2012.
[5] Hertwich EG, Gibon T, Bouman EA, Arvesen A, Suh S, Heath GA, et al. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. Proceedings of the National Academy of Sciences. 2014;10.1073/pnas.1312753111.

Appendix C: Paper III

Bouman, E. A., C. Skar, and E. G. Hertwich. 2015c. LCA of electricity technologies using capacity factors dependent on economic dispatch. *Submitted to Environmental Research Letters*

Is not included due to copyright

Appendix D: Paper IV

Bouman, E. A., C. Skar, and E. Hertwich. 2015d. Specific renewable energy technology targets can reduce life cycle impacts of electricity generation. *Submitted to Environmental Science & Technology*

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Appendix E: Supporting Paper I

Hertwich, E. G., T. Gibon, E. A. Bouman, A. Arvesen, S. Suh, G. A. Heath, J. D. Bergesen, A. Ramirez, M. I. Vega, and L. Shi. **2015**. Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies. Proceedings of the National Academy of Sciences 112(20): 6277-6282.

Published version of the paper available at:
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SPECIAL FEATURE

Integrated life-cycle assessment of electricity-supply scenarios confirms global environmental benefit of low-carbon technologies

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Decarbonization of electricity generation can support climate-change mitigation and presents an opportunity to address pollution resulting from fossil-fuel combustion. Generally, renewable technologies require higher initial investments in infrastructure than fossil-based power systems. To assess the tradeoffs of increased up-front emissions and reduced operational emissions, we present, to our knowledge, the first global, integrated life-cycle assessment (LCA) of long-term, wide-scale implementation of electricity generation from renewable sources (i.e., photovoltaic and solar thermal, wind, and hydropower) and of carbon dioxide capture and storage for fossil power generation. We compare emissions causing particulate matter exposure, freshwater ecotoxicity, freshwater eutrophication, and climate change for the climate-change-mitigation (BLUE Map) and business-as-usual (Baseline) scenarios of the International Energy Agency up to 2050. We use a vintage stock model to conduct an LCA of newly installed capacity year-by-year for each region, thus accounting for changes in the energy mix used to manufacture future power plants. Under the Baseline scenario, emissions of air and water pollutants more than double whereas the low-carbon technologies introduced in the BLUE Map scenario allow a doubling of electricity supply while stabilizing or even reducing pollution. Material requirements per unit generation for low-carbon technologies can be higher than for conventional fossil generation: 11–40 times more copper for photovoltaic systems and 6–14 times more iron for wind power plants. However, only two years of current global copper and one year of iron production will suffice to build a low-carbon energy system capable of supplying the world's electricity needs in 2050.

land use | climate-change mitigation | air pollution | multiregional input–output | CO_2 capture and storage

A shift toward low-carbon electricity sources has been shown to be an essential element of climate-change mitigation strategies (1, 2). Much research has focused on the efficacy of technologies to reduce climate impacts and on the financial costs of these technologies (2–4). Some life-cycle assessments (LCAs) of individual technologies suggest that, per unit generation, low-carbon power plants tend to require more materials than fossil-fueled plants and might thereby lead to the increase of some other environmental impacts (5, 6). However, little is known about the environmental implications of a widespread, global shift to a low-carbon electricity supply infrastructure. Would the material and construction requirements of such an infrastructure be large relative to current production capacities? Would the shift to low-carbon electricity systems increase or decrease other types of pollution? Energy-scenario models normally do not represent the manufacturing or material life cycle of energy technologies and are therefore not capable of answering such questions. LCAs typically address a single technology at a time. Comparative studies often focus on a single issue, such as selected pollutants (7), or the use of land (8) or metals (9, 10). They do not trace the interaction between different technologies. Existing comparative analyses are based on disparate, sometimes outdated literature data (7, 11, 12), which raises issues regarding differences in assumptions, system boundaries, and input data, and therefore the comparability and reliability of the results. Metaanalyses of LCAs address some of these challenges (13, 14), but, to be truly consistent, a comparison of technologies should be conducted within a single analytical structure, using the same background data for common processes shared among technologies, such as component materials and transportation. The benefits of integrating LCA with other modeling approaches, such as input–output analysis, energy-scenario modeling, and material-flow analysis have been suggested in recent reviews (7, 15).

We analyze the environmental impacts and resource requirements of the wide-scale global deployment of different low-carbon electricity generation technologies as foreseen in one prominent climate-change mitigation scenario [the International Energy Agency's (IEA) BLUE Map scenario], and we compare it with the IEA's Baseline scenario (16). To do so, we developed an integrated hybrid LCA model that considers utilization of the selected energy technologies in the global production system and includes several efficiency improvements in the production system assumed in the BLUE Map scenario. This model can

Significance

Life-cycle assessments commonly used to analyze the environmental costs and benefits of climate-mitigation options are usually static in nature and address individual power plants. Our paper presents, to our knowledge, the first life-cycle assessment of the large-scale implementation of climate-mitigation technologies, addressing the feedback of the electricity system onto itself and using scenario-consistent assumptions of technical improvements in key energy and material production technologies.

Author contributions: E.G.H., T.G., and S.S. designed research; E.G.H., T.G., E.A.B., A.A., S.S., G.A.H., J.D.B., A.R., M.I.V., and L.S. performed research; T.G., E.A.B., A.A., and J.D.B. contributed new reagents/analytic tools; E.G.H., T.G., E.A.B., A.A., and J.D.B. analyzed data; and E.G.H., T.G., E.A.B., A.A., S.S., G.A.H., J.D.B., A.R., M.I.V., and L.S. wrote the paper.

The authors declare no conflict of interest.

This article is a PNAS Direct Submission.

Data deposition: The life-cycle inventory data are available on the Norwegian University of Science and Technology website, www.ntnu.no/documents/10370/1021067956/Environmental+assessment+of+clean+electricity.

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This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1312753111/-/DCSupplemental.

SUSTAINABILITY SCIENCE

ENVIRONMENTAL SCIENCES

address the feedback of the changing electricity mix on the production of the energy technologies.

We collected original life-cycle inventories for concentrating solar power (CSP), photovoltaic power (PV), wind power, hydropower, and gas- and coal-fired power plants with carbon dioxide (CO_2) capture and storage (CCS) according to a common format, and we provide these inventories in *SI Appendix*. Bioenergy was excluded because an assessment would require a comprehensive assessment of the food system, which was beyond the scope of this work. Nuclear energy was excluded because we could not reconcile conflicting results of competing assessment approaches (17). To reflect the prospective nature of our inquiry, the modeling of technologies implemented in 2030 and 2050 also contains several assumptions regarding the improved production of aluminum, copper, nickel, iron and steel, metallurgical grade silicon, flat glass, zinc, and clinker (18). These improvements represent an optimistic-realistic development in accordance with predictions and goals of the affected industries, as specified in ref. 18 and summarized in *SI Appendix*, Table S1. Technological progress in the electricity conversion technologies was represented through improved conversion efficiencies, load factors, and next-generation technology adoption to achieve the technology performance of the scenarios (see *SI Appendix* for details).

Results has two parts. First, low-carbon technologies are compared with fossil electricity generation without CCS to quantify environmental cobenefits and tradeoffs relevant for long-term investment decisions in the power sector. This comparison reflects the current state-of-the-art technology performance for both low-carbon and fossil systems. We examine impacts in terms of greenhouse gas (GHG) emissions, eutrophication, particulate-matter formation, and aquatic ecotoxicity resulting from pollutants emitted to air and water throughout the life cycle of each technology. We also compare the life-cycle use of key materials (namely aluminum, iron, copper, and cement), nonrenewable energy, and land for all investigated technologies per unit of electricity produced. *SI Appendix* contains a discussion of technology-specific results. To our knowledge, this analysis is the first to be based on a life-cycle inventory model that includes the feedback of the changing electricity mix and the effects of improvements in background technologies on the production of the energy technologies.

In the second part of *Results*, we show the potential resource requirements and environmental impacts of the evaluated technologies within the BLUE Map scenario and compare these results with those of the Baseline scenario. Our modeling is based on the installation of new capacity and the utilization of this capacity such that it is consistent with the BLUE Map scenario. It traces an important aspect of the transition toward a low-carbon energy system: that new capacity of low-carbon electricity generation technology is constructed using the existing electricity mix at any point of time. We quantify the requirements of bulk materials and the environmental pressures associated with the BLUE Map scenario over time and compare them with the Baseline scenario. We then compare results to annual production levels of these materials. In *Discussion*, we examine issues related to the presented work, in particular the implication of life-cycle effects on the modeling of mitigation scenarios and limitations with respect to the grid integration of variable renewable supply.

Results

Technology Comparison per Unit Generation. Our comparative LCA indicates that renewable energy technologies have significantly lower pollution-related environmental impacts per unit of generation than state-of-the-art coal-fired power plants in all of the impact categories we consider (Fig. 1 and *SI Appendix*, Table S5). Modern natural gas combined cycle (NGCC) plants could also cause very little eutrophication, but they tend to lie between renewable technologies and coal power for climate change (Fig. 1*A*) and ecotoxicity (Fig. 1*C*). NGCC plants also have higher contributions of particulate matter exposure (Fig. 1*B*). The LCA finds that wind and solar power plants tend to require more bulk materials (namely, iron, copper, aluminum, and cement) than coal- and gas-based electricity per unit of generation (Fig. 1 *G–J*). For fossil fuel-based power systems, materials contribute a small fraction to total environmental impacts, corresponding to <1% of GHG emissions for systems without CCS and 2% for systems with CCS. For renewables, however, materials contribute 20–50% of the total impacts, with CSP tower and offshore wind technologies showing the highest shares (*SI Appendix*, Fig. S1). However, the environmental impact of the bulk material requirements of renewable technologies (*SI Appendix*, Table S1) is still small in absolute terms compared with the impact of fuel production and combustion of fossil-based power plants (Fig. 1).

CCS reduces CO_2 emissions of fossil fuel-based power plants but increases life-cycle indicators for particulate matter, ecotoxicity, and eutrophication by 5–60% (Fig. 1 *B–D*). Both postcombustion and precombustion CCS require roughly double the materials of a fossil plant without CCS (Fig. 1 *G–J*). The carbon capture process itself requires energy and therefore reduces efficiency, explaining much of the increase in air pollution and material requirements per unit of generation.

Habitat change is an important cause of biodiversity loss (19). Habitat change depends both on the project location and on the specific area requirement of the technology. For example, PV power may be produced in pristine natural areas (high impact on habitat) or on rooftops (low impact on habitat). A detailed assessment of specific sites used for future power plants is beyond the scope of this global assessment. As an indicator of potential habitat change, we use the area of land occupied during the life cycle of each technology (Fig. 1*E*).

High land-use requirements are associated with hydropower reservoirs, coal mines, and CSP and ground-mounted PV power plants. The lowest land use requirements are for NGCC plants, wind, and roof-mounted PV. We consider roof-mounted PV to have zero direct land use because the land is already in use as a building. For ground-mounted solar power, we consider the entire power plant because the modules or mirrors are so tightly spaced that agriculture and other uses are not feasible in the unoccupied areas. Considering only the space physically occupied by the installation, the area requirements decrease by a factor of 2–3 compared with the values in Fig. 1*E* (8). For direct land use associated with wind power, we consider only the area occupied by the wind turbine itself, access roads, and related installations. We do not include the land between installations because it can be used for other purposes such as agriculture or wilderness, with some restrictions (20). If an entire land-based wind park is considered, land use would be on the order of 50–200 square meter-year/MWh (m^2a/MWh) (8, 20), which is higher than other technologies. We do not account for the use of sea area by offshore wind turbines.

Cumulative nonrenewable (fossil or nuclear) energy consumption is of interest because it traces the input of a class of limited resources. The current technologies used in the production of renewable systems consume 0.1–0.25 kWh of nonrenewable energy for each kWh of electricity produced (Fig. 1*F*). The situation is different for fossil fuel-based systems, for which the cumulative energy consumption reflects the efficiency of power production and the energy costs of the fuel chain and, if applicable, the CCS system.

Scenario Results. The BLUE Map scenario posits an increase in the combined share of solar, wind, and hydropower from 16.5% of total electricity generation in 2010 to 39% in 2050. The required up-front investment in renewable generation capacity

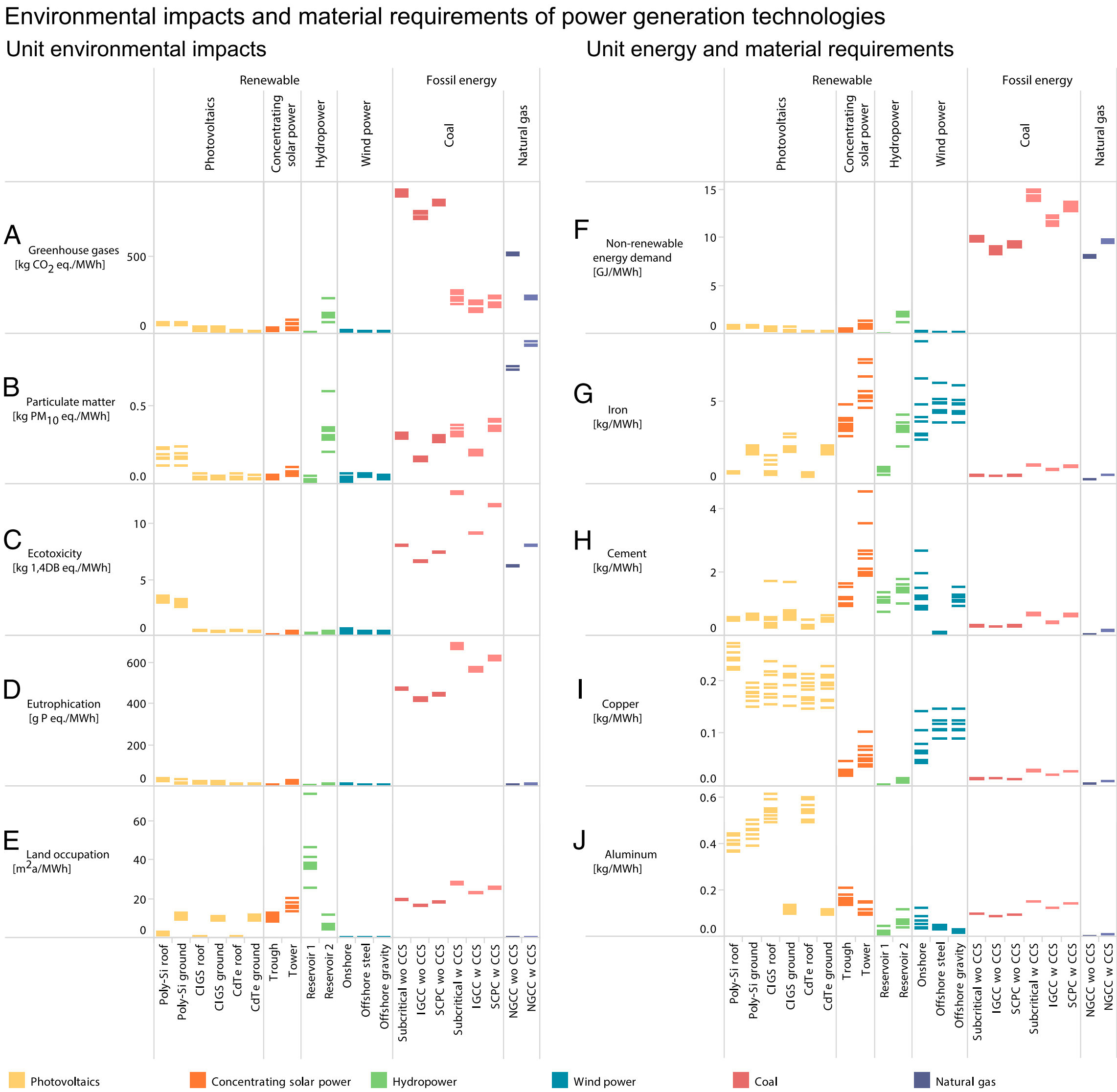

Fig. 1. A comparison of life-cycle environmental pressures and resource use per unit of electricity generated by different power-generation technologies in each of nine world regions. The left column shows four pollution-oriented indicators: (*A*) Greenhouse gases, (*B*) particulate matter exposure, (*C*) freshwater ecotoxicity, and (*D*) freshwater eutrophication. In addition, land occupation (*E*) is shown. The right column indicates nonrenewable primary energy demand (*F*) and the demand for materials (*G*–*J*). CCS, CO_2 capture and storage; CdTe, cadmium telluride; CIGS, copper indium gallium selenide; IGCC, integrated gasification combined cycle coal-fired power plant; NGCC, natural gas combined cycle power plant; offshore gravity, offshore wind power with gravity-based foundation; offshore steel, offshore wind power with steel-based foundation; reservoir 2, type of hydropower reservoir used as a higher estimate; SCPC, supercritical pulverized coal-fired power plant.

would require a combined investment of bulk materials of 1.5 Gt over the period 2010–2050, which is more than the total use of these materials in the Baseline scenario. Because of the need to install new renewable capacity, the material requirement of the BLUE Map scenario is from the outset higher than that of the Baseline scenario, even as the generation profiles are initially quite similar. The difference in material demand displayed in Fig. 2 *G*–*J* shows that the initial demand for iron and cement is mainly associated with wind and CSP installations whereas it is mainly PV driving additional copper demand. The BLUE Map scenario has a lower material demand associated with conventional coal-fired power plants without CCS, which is partly offset by the material demand from coal-fired power plants with CCS. The most important contributor to the material demand from coal-fired power plants is associated with producing and transporting the ∼500 kg of coal required per MWh of electricity generated.

The BLUE Map scenario would be able to keep the emissions of particulate matter and ecotoxicity stable despite the doubling of annual electricity generation from 18 petawatt hours per

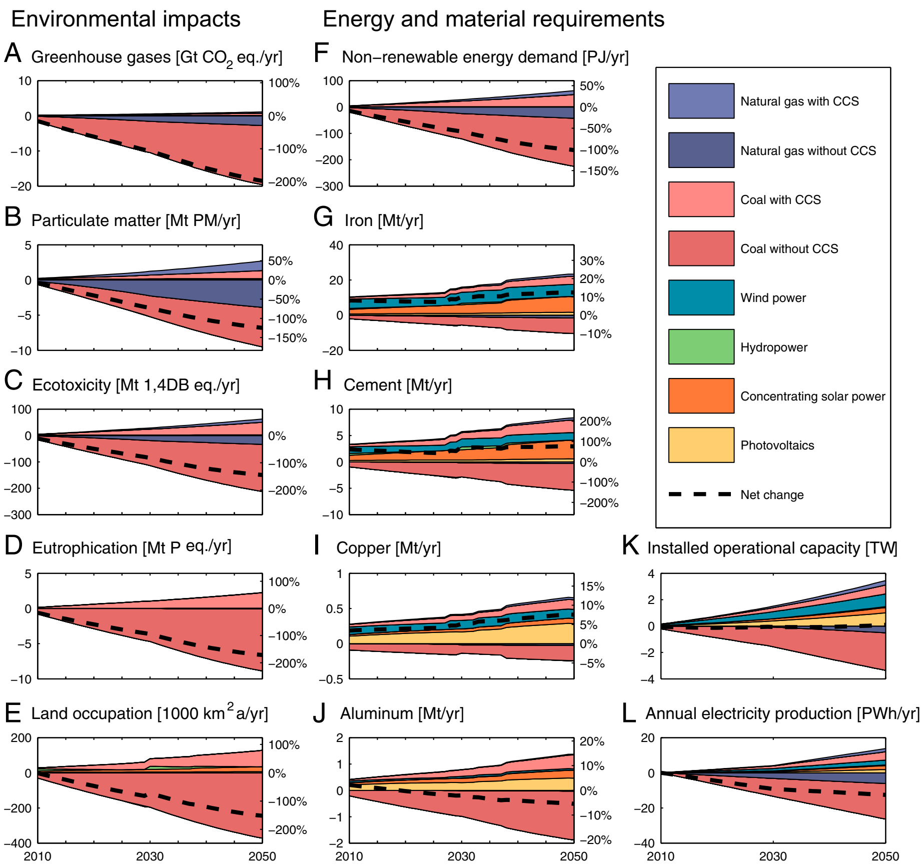

Fig. 2. (*A–L*) Environmental and resource implications of electricity generation following the IEA BLUE Map scenario instead of the IEA Baseline scenario, addressing impacts from the indicated power sources. The results show a reduction of pollution-related environmental impacts despite a doubling of electricity generation but a substantial increase of material consumption, especially copper. Left axes show absolute values. Right axes show the variation, in percentage, between these absolute values and the base levels in 2007. Note that the net change can reach values below −100% when the difference between the Baseline and BLUE Map scenarios is higher than the base 2007 levels.

annum (PWh/a) to 36 PWh/a for the technologies investigated. Compared with the situation in 2010, a substantial reduction in GHG emissions (from 9.4 Gt CO_2 eq. to 3.4 Gt CO_2 eq.) and eutrophication would be achieved (*SI Appendix*, Fig. S4). In stark contrast, the Baseline scenario would lead to a doubling of all pollution-related indicators even as new, highly efficient coal-fired power plants come online (*SI Appendix*, Fig. S3). The difference in pollution between the BLUE Map and Baseline scenarios would grow dramatically over time (Fig. 2) whereas the additional required material investment would rise only moderately. Such a development is the result of the growing dividend from the continuous investment in renewable generation capacity.

For the BLUE Map scenario, the higher material requirement per unit of renewable electricity and a projected increase in energy demands cause a substantial increase in material use (*SI Appendix*, Fig. S4). The overall material requirement per unit of electricity produced would be 2.3 kg/MWh compared with 1.2 kg/MWh for the Baseline scenario. That increase appears manageable in the context of current production volumes, the long lifetime of the equipment, and the ability to recycle the metals. Compared with material production levels in 2011, the construction and operation of the 2050 electricity system envisioned in the BLUE Map scenario would require less than 20% of the cement, 90% of the iron, 150% of the aluminum, and 200% of the copper, all relative to their respective 2011 production quantities (Table 1). Meeting copper demand could be problematic due to declining ore grades (21), and it would result in potential increases in the environmental costs of copper production (22, 23). Additional evidence for this conclusion is presented in *SI Appendix*.

Displacing fossil fuels through the widespread deployment of solar and wind energy could limit air and water pollution (Fig. 2). Over the study period (2010–2050), emissions of GHG connected to the power plants investigated are 62% lower in BLUE Map than they are in the Baseline Scenario whereas the particulate matter is 40% lower, freshwater ecotoxicity is almost 50% lower, and eutrophication is 55% lower. Furthermore, both cumulative energy consumption and land use are reduced. Our analysis might understate the cobenefits of climate-change mitigation in the form of pollution reduction because we assume the replacement of state-of-the-art fossil power plants with well-operating, modern emissions control equipment; the actual situation might be that emissions control equipment are functioning suboptimally or are altogether absent due to a lack of regulation.

Table 1. Cumulative material requirements for electricity production for the BLUE Map scenario

Material	Annual production (2011), Gt	Metal requirements to 2050, Gt	Ratio
Aluminum	0.045	0.067	1.5
Copper	0.013	0.029	2.2
Iron	1.5	1.3	0.87
Cement	3.4	0.52	0.15

The middle column provides an estimate of the volumes of materials that need to be produced to provide for the capital stock additions between 2010 and 2050 and the material requirements associated with operational inputs (fuels, transport, solvents, etc.) during the same period. The right hand column expresses these material requirements as a fraction of the 2011 production volume.

Further results on specific technologies, GHG emissions from material production, and the scenario analysis are presented in *SI Appendix*.

Discussion

Previous assessments of life-cycle impacts of electricity-generation technologies have used static LCAs (7, 11–15). Technologies are thus analyzed side-by-side, assuming current production technologies. We present an assessment based on an integrated, scenario-based hybrid LCA model with global coverage through the integration of the life-cycle process description in a nine-region multiregional input–output model. Integration of the life-cycle model, in which new technologies become part of the electricity mix and thus the life cycle of the same and other new technologies, addresses the interaction among technologies. Adopting a vintage capital model, the life-cycle stages of individual power plants are explicitly in time, also a novelty compared with current LCA practice. This previously unidentified type of modeling approach thus provides the ability to model the role of various technologies in a collectively exhaustive and mutually exclusive way. Only through this integration can the life-cycle emissions and resource use of energy scenarios be analyzed correctly. Further, we can assess the contributions of changes in the technology mix and improvements in the technology itself to future reductions of environmental impacts, as demonstrated in ref. 24.

The widespread utilization of variable sources such as solar and wind energy raises the question: what are the additional environmental costs of matching supply and demand? Grid-integration measures for variable supply, such as the stand-by operation of fossil fuel power plants, grid expansion, demand-response and energy storage (25–27), result in extra resource requirements and environmental impacts (28). The challenges of balancing supply and demand are not yet severe in the BLUE Map scenario, in which variable wind and solar technologies cover 24% of the total electricity production in 2050, but balancing becomes a serious concern later in the century in the many mitigation scenarios investigated by ref. 2 that rely on a higher share of variable renewables. In the BLUE Map scenario, the capacity factor of fossil fuel-fired power plants without CCS is reduced from 40% in 2007 to 19% in 2050 for natural gas, and from 65% to 30% for coal for the same period, but IEA provides no information on emissions associated with spinning reserves, or ramp-up and ramp-down. The National Renewable Energy Laboratory's (NREL) Western Wind and Solar Integration Study indicates that increased fossil power plant cycling from the integration of a similar share of variable renewables may result in only negligible increases in greenhouse gas emissions compared with a scenario without renewables. It may also result in further reductions in nitrogen oxide emissions and increases in SO_2 emissions equal to about 2–5% of the total emissions reduced by using renewables. In a study investigating an 80% emission reduction in California, electricity storage requirements become significant only at higher rates of renewable energy penetration (26). See *SI Appendix* for further information on grid integration of renewables. Additional research on different options for the system integration of renewables and its environmental impact is required to determine the share of renewables most desirable from an environmental perspective.

Our analysis raises important questions. (*i*) What would similar analyses of other mitigation scenarios look like? Thousands of scenarios have been collected in the Intergovernmental Panel on Climate Change (IPCC) mitigation scenario analysis database (4). These scenarios use a combination of energy conservation, renewable and nuclear energy, and CCS. Our analysis suggests that an electricity supply system with a high share of wind energy, solar energy, and hydropower would lead to lower environmental impacts than a system with a high share of CCS. (*ii*) How can scenarios for a wider range of environmental impacts be routinely assessed? Endogenous treatment of equipment life cycles as considered here in energy-scenario models has not yet been achieved. Options are either to (*a*) include some simplified assessments in energy scenario models, using the unit-based results from our analysis in the scenario models, or to (*b*) conduct a postprocessing of scenario results in the manner done for this study. The advantage of option *a* is that life-cycle emissions could be considered in the scenario development, thus affecting the technology choice; the advantage of option *b* is the ability to include feedbacks and economy-wide effects in the calculation of life-cycle emissions. (*iii*) Will fundamental differences in energy systems such as those between mitigation and baseline scenarios lead to significant changes to the supply and demand for many products (e.g., fuels and raw materials)? It is clear that there will be effects on the supply and demand of goods both due to different energy policies (e.g., carbon prices) and because of differences in the demand and supply of resources (e.g., iron or coal) to the global economy. Such indirect effects were outside of the scope of this study, but they could be considered in a consequential analysis (29).

Conclusions

Our analysis indicates that the large-scale implementation of wind, PV, and CSP has the potential to reduce pollution-related environmental impacts of electricity production, such as GHG emissions, freshwater ecotoxicity, eutrophication, and particulate-matter exposure. The pollution caused by higher material requirements of these technologies is small compared with the direct emissions of fossil fuel-fired power plants. Bulk material requirements appear manageable but not negligible compared with the current production rates for these materials. Copper is the only material covered in our analysis for which supply may be a concern.

Materials and Methods

Using a uniform data-collection form, we collected foreground data describing the life-cycle inventory of the analyzed technologies. For more information on inventory data and modeling assumptions, see *SI Appendix*. These foreground data were linked to the ecoinvent 2.2 life-cycle inventory database (30), which provides information on many input processes such as materials and manufacturing, and the EXIOBASE input–output database (31), which provides emissions estimates for inputs of services and highly manufactured goods. We modeled nine world regions to perform a regional sensitivity analysis. Exogenous scenario parameters and electricity mixes were taken from the IEA scenarios (16), which represent the same nine world regions. Impact assessment was conducted using ReCiPe version 1.08 (32). To specify resource use, cumulative nonrenewable energy demand, land use, and the use of iron, aluminum, and copper (metal content of the ore or scrap used) were specified. To complement environmentally important material flows (33), we also quantified the amount of cement required. Life-cycle inventories for this comparative analysis were built based on our original work and a review of scientific literature on the selected technologies. To obtain a better representation of the fugitive methane emissions related to fossil-fuel extraction, ecoinvent 2.2 was updated with the fugitive emissions factors published in ref. 34, which is in line with other recent estimates.

To develop the scenarios of emissions and resource use presented in Fig. 2 and *SI Appendix*, Figs. S3 and S4, we identified the timing of capacity additions, operations, repowering, and removal of power plants in the scenario (35). We delineated the life-cycle impacts into these phases. Therefore, the figures reflect the timing of resource use and emissions, not the timing of electricity generation. The inventories associated with each life cycle step reflect the technology status and electricity mix of the year in question. The IEA provides electricity production by technology group (e.g., PV), so we estimated intratechnology group market shares [e.g., the division of the PV market among Si, cadmium telluride (CdTe), and copper indium gallium selenide (CIGS) technologies]. As of 2010, 90% of the PV market in terms of produced electricity was silicon-based whereas the remaining share consisted of thin-film modules. The share of silicon-based modules gradually decreases to 20% in 2050. Half of the electricity produced by CSP was assumed to be generated from central receivers systems; the other half was assumed to be from parabolic troughs. This allocation remained consistent throughout the scenario time frame. Hydropower plants were represented by two different dams modeled after the Baker River Basin dams in Chile. Unit results show high variability, even within the same river basin. Wind power plants were assumed to contain conventional gearbox-equipped wind turbines because reliable LCA data on rare earth metal use in direct drive wind turbines could not be obtained. Offshore wind farm production was modeled as an even mix of gravity-based and steel foundation turbines. The market mix of coal-combustion technologies was modeled after real production data for China, India, and the United States. A global average was applied for other regions. Due to high uncertainty of coal market share estimates, we used the 2010 mix for 2030 and 2050. We assumed all gas-fired power plants used combined cycle technology.

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1. Riahi K, et al. (2012) Energy pathways for sustainable development. *Global Energy Assessment: Toward a Sustainable Future* (Cambridge Univ Press, Cambridge, UK), pp 1203–1306.
2. Bashmakov IA, et al. (2014) Energy systems. *Climate Change 2014: Mitigation of Climate Change*, ed Edenhofer O (Intergovernmental Panel on Climate Change, Geneva).
3. Intergovernmental Panel on Climate Change (2005) *Carbon Dioxide Capture and Storage: Special Report of the Intergovernmental Panel on Climate Change* (Cambridge Univ Press, Cambridge, UK).
4. Intergovernmental Panel on Climate Change (2011) *Special Report on Renewable Energy Sources and Climate Change Mitigation* (Cambridge Univ Press, Cambridge, UK).
5. Singh B, Strømman AH, Hertwich EG (2011) Comparative life cycle environmental assessment of CCS technologies. *Int J Greenh Gas Control* 5(4):911–921.
6. Arvesen A, Hertwich EG (2012) Assessing the life cycle environmental impacts of wind power: A review of present knowledge and research needs. *Renew Sustain Energy Rev* 16(8):5994–6006.
7. Sathaye J, et al. (2011) Renewable energy in the context of sustainable energy. *IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation*, eds Edenhofer O, et al. (Cambridge Univ Press, Cambridge, UK).
8. Fthenakis V, Kim HC (2009) Land use and electricity generation: A life-cycle analysis. *Renew Sustain Energy Rev* 13(6-7):1465–1474.
9. Alonso E, et al. (2012) Evaluating rare earth element availability: A case with revolutionary demand from clean technologies. *Environ Sci Technol* 46(6):3406–3414.
10. Kleijn R, van der Voet E, Kramer GJ, van Oers L, van der Giesen C (2011) Metal requirements of low-carbon power generation. *Energy* 36(9):5640–5648.
11. Jacobson MZ (2009) Review of solutions to global warming, air pollution, and energy security. *Energy Environ. Sci.* 2:148–173.
12. Weisser D (2007) A guide to life-cycle greenhouse gas (GHG) emissions from electric supply technologies. *Energy* 32(9):1543–1559.
13. Heath GA, Mann MK (2012) Background and reflections on the Life Cycle Assessment Harmonization Project. *J Ind Ecol* 16(Suppl 1):S8–S11.
14. Brandão M, Heath G, Cooper J (2012) What can meta-analyses tell us about the reliability of life cycle assessment for decision support? *J Ind Ecol* 16(Suppl 1):S3–S7.
15. Masanet E, et al. (2013) Life-cycle assessment of electric power systems. *Annu Rev Environ Resour* 38(1):107–136.
16. International Energy Agency (2010) Energy technology perspectives 2010: Scenarios and strategies to 2050 (Organisation for Economic Co-operation and Development/ International Energy Agency, Paris).
17. Warner ES, Heath GA (2012) Life cycle greenhouse gas emissions of nuclear electricity generation: Systematic review and harmonization. *J Ind Ecol* 16(Suppl 1):S73–S92.
18. ESU Services, Institut für Energie- und Umweltforschung (2008) LCA of background processes. Deliverable D15.1, New Energy Externalities Developments for Sustainability (NEEDS) Integrated Project. (European Union Sixth Framework Programme, Brussels).
19. Mooney HA, et al., eds (2005) *Ecosystems and Human Well-Being: Synthesis* (Island Press, Washington, DC).
20. Denholm P, Hand M, Jackson M, Ong S (2009) *Land-Use Requirements of Modern Wind Power Plants in the United States* (National Renewable Energy Laboratory, Golden, CO).
21. Gordon RB, Bertram M, Graedel TE (2006) Metal stocks and sustainability. *Proc Natl Acad Sci USA* 103(5):1209–1214.
22. van der Voet E, et al. (2012) Environmental challenges of anthropogenic metals flows and cycles. *IRP Report*, ed International Resource Panel (United Nations Environment Programme, Nairobi, Kenya).
23. Norgate T, Haque N (2010) Energy and greenhouse gas impacts of mining and mineral processing operations. *J Clean Prod* 18(3):266–274.
24. Bergesen JD, Heath GA, Gibon T, Suh S (2014) Thin-film photovoltaic power generation offers decreasing greenhouse gas emissions and increasing environmental co-benefits in the long term. *Environ Sci Technol* 48(16):9834–9843.
25. Kempton W, Pimenta FM, Veron DE, Colle BA (2010) Electric power from offshore wind via synoptic-scale interconnection. *Proc Natl Acad Sci USA* 107(16):7240–7245.
26. Williams JH, et al. (2012) The technology path to deep greenhouse gas emissions cuts by 2050: The pivotal role of electricity. *Science* 335(6064):53–59.
27. Göransson L, Johnsson F (2011) Large scale integration of wind power: Moderating thermal power plant cycling. *Wind Energy (Chichester Engl)* 14(1):91–105.
28. Pehnt M, Oeser M, Swider DJ (2008) Consequential environmental system analysis of expected offshore wind electricity production in Germany. *Energy* 33(5):747–759.
29. Zamagni A, Guinée J, Heijungs R, Masoni P, Raggi A (2012) Lights and shadows in consequential LCA. *Int J Life Cycle Assess* 17(7):904–918.
30. Ecoinvent (2007) Ecoinvent 2.0 (Swiss Centre for Life Cycle Inventories, Dübendorf, Switzerland).
31. Tukker A, et al. (2013) EXIOPOL: Development and illustrative analyses of detailed global multiregional, environmentally extended supply and use tables and symmetric input-output tables. *Econ Syst Res* 25(1):50–70.
32. Goedkoop M, et al. (2008) Report I: Characterisation. *ReCiPe 2008: A Life Cycle Impact Assessment Method Which Comprises Harmonised Category Indicators at the Midpoint and the Endpoint Level* (Dutch Ministry of the Environment, The Hague, The Netherlands), 1st Ed.
33. Allwood JM, Cullen JM, Milford RL (2010) Options for achieving a 50% cut in industrial carbon emissions by 2050. *Environ Sci Technol* 44(6):1888–1894.
34. Burnham A, et al. (2012) Life-cycle greenhouse gas emissions of shale gas, natural gas, coal, and petroleum. *Environ Sci Technol* 46(2):619–627.
35. Arvesen A, Hertwich EG (2011) Environmental implications of large-scale adoption of wind power: A scenario-based life cycle assessment. *Environ Res Lett* 6(4):045102.