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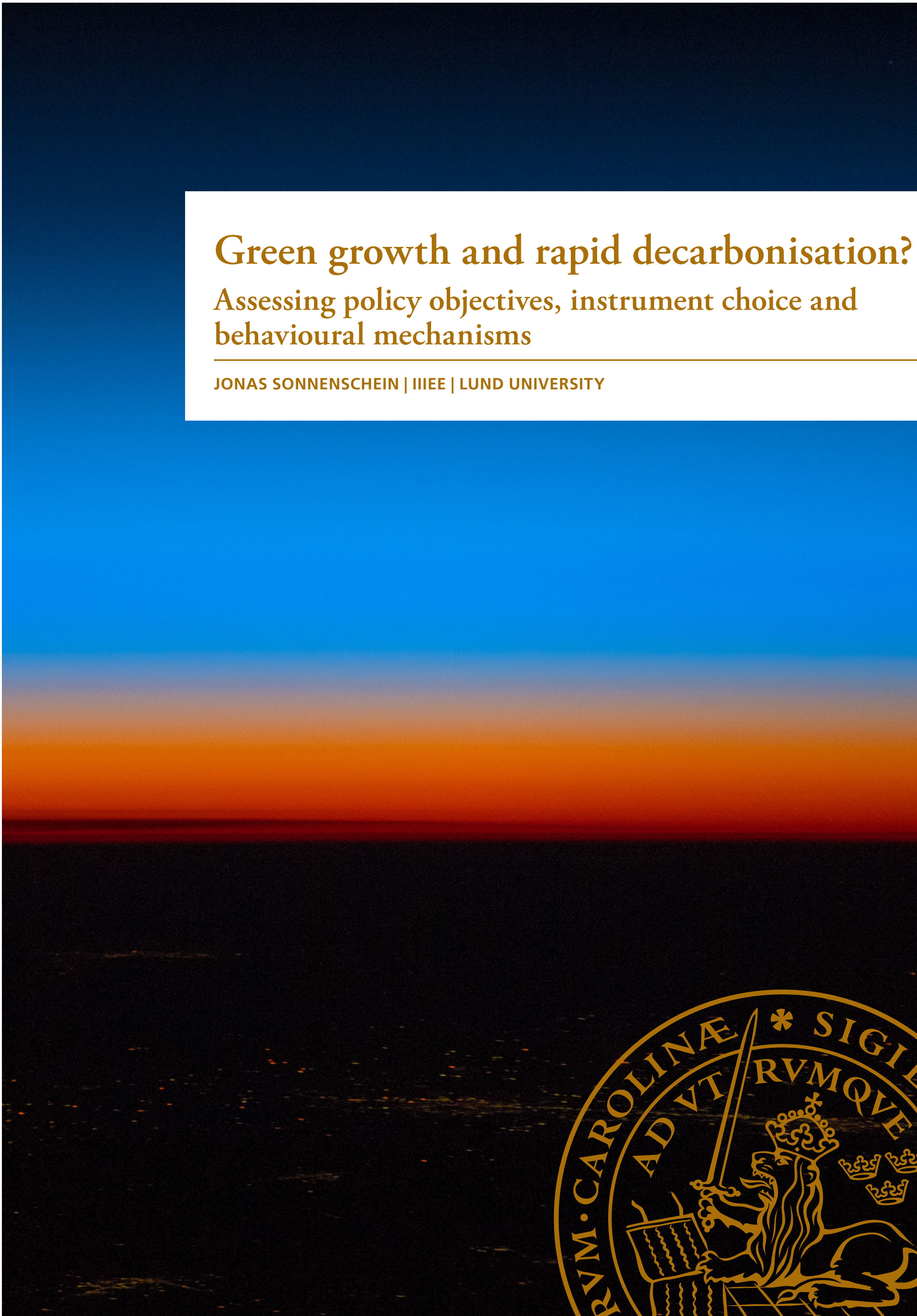
Sonnenschein, Jonas



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JONAS SONNENSCHEIN | IIIEE | LUND UNIVERSITY

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Jonas Sonnenschein

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Abstract
After the 2008/2009 global financial crisis, significant policy and scientific attention has been given to 'green growth', which aims at the concurrent achievement of economic growth and GHG emissions reductions via de-coupling. Despite ambitious aims, there is a lack of empirical knowledge about the effects and the potential of green growth climate mitigation policies.

This thesis aims at providing new policy-relevant understanding of how green growth climate mitigation policies can contribute towards rapid decarbonisation. The specific policies assessed in this thesis include green fiscal stimulus in the Republic of Korea, public financing for the development of low-carbon energy technologies in the Nordic countries, carbon pricing and minimum energy performance standards for electric appliances in the UK, and carbon pricing mechanisms for personal transport in Sweden with a focus on air ticket taxes. Methodologically, the thesis presents policy assessments that deploy a variety of mainly quantitative research methods. The analysis goes beyond short-term cost-effectiveness and takes into consideration a more behaviourally realistic model of decision-making in response to economic policy instruments.

It is found that green growth climate policies have not yet driven the short- to mid-term decarbonisation needed to reach the targets of the Paris Agreement. This is explained by the strong economic growth objectives of green growth climate policies, by insufficient policy-stringency, and by the disregard of behavioural mechanisms (potentially) affecting policy outcomes. Regulatory policies may offer an effective alternative to economic incentives (carbon pricing in particular) in order to achieve a mitigation effect that is consistent with the targets of the Paris Agreement. Careful framing and targeting of carbon pricing can, however, increase its emissions reduction potential, and several behavioural factors are identified that may increase its policy acceptance and effectiveness. These findings confirm that a more integrated policy assessment approach is needed to support the design and implementation of green growth climate mitigation policies.

The thesis underlines several policy implications. It shows that the simultaneous achievement of both economic and climate objectives is difficult in practice, particularly when policies are not stringent enough. Whereas innovations in low-carbon energy technologies are critical for decarbonisation, behavioural aspects concerning the adoption of LCET and sustainable energy use are equally important. Well-designed carbon pricing, both explicit and implicit, should be an important element of the green growth climate policy mix.

Key words climate change, mitigation, policy, carbon pricing, green growth, market failure, behavioural insights

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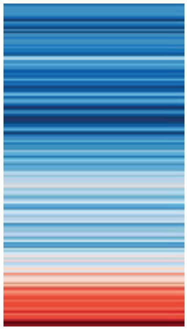
The cover photo entitled ‘Californian fires’ was taken by ISS astronaut Alexander Gerst in November 2018. He wrote about it: ‘Looking down on countless fires when flying over California. Tough to see so much destruction, and we can’t do anything about it from up here.’ To me the picture illustrates how vulnerable the atmosphere is. It also shows that impacts of climate change can already be seen from space. The colour spectrum of the sky reminds me of the ‘climate stripes’ illustration of global average temperature development between 1850 and 2017 by Ed Hawkins, pictured to the right.



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MADE IN SWEDEN

To the polar bear, the coral reef and the rebel

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Finally, a big warm 'Thank you!' to all my friends and family! Joana and Murat, thanks for providing me a home away from home! To my parents, thank you for your encouragement, support and trust, which gave me the opportunity to go out into the world, study and eventually write this thesis! And without you, Anja, Mila and Katja, life outside of work would not be half as fun. You make my day. Every day. Thank you!

Jonas
Ljubljana, April 2019

Abstract

After the 2008/2009 global financial crisis, significant policy and scientific attention has been given to 'green growth', which aims at the concurrent achievement of economic growth and GHG emissions reductions via de-coupling. Despite ambitious aims, there is a lack of empirical knowledge about the effects and the potential of green growth climate mitigation policies.

This thesis aims at providing new policy-relevant understanding of how green growth climate mitigation policies can contribute towards rapid decarbonisation. The specific policies assessed in this thesis include green fiscal stimulus in the Republic of Korea, public financing for the development of low-carbon energy technologies in the Nordic countries, carbon pricing and minimum energy performance standards for electric appliances in the UK, and carbon pricing mechanisms for personal transport in Sweden with a focus on air ticket taxes. Methodologically, the thesis presents policy assessments that deploy a variety of mainly quantitative research methods. The analysis goes beyond short-term cost-effectiveness and takes into consideration a more behaviourally realistic model of decision-making in response to economic policy instruments.

It is found that green growth climate policies have not yet driven the short- to mid-term decarbonisation needed to reach the targets of the Paris Agreement. This is explained by the strong economic growth objectives of green growth climate policies, by insufficient policy-stringency, and by the disregard of behavioural mechanisms (potentially) affecting policy outcomes. Regulatory policies may offer an effective alternative to economic incentives (carbon pricing in particular) in order to achieve a mitigation effect that is consistent with the targets of the Paris Agreement. Careful framing and targeting of carbon pricing can, however, increase its emissions reduction potential, and several behavioural factors are identified that may increase its policy acceptance and effectiveness. These findings confirm that a more integrated policy assessment approach is needed to support the design and implementation of green growth climate mitigation policies.

The thesis underlines several policy implications. It shows that the simultaneous achievement of both economic and climate objectives is difficult in practice, particularly when policies are not stringent enough. Whereas innovations in low-carbon energy technologies are critical for decarbonisation, behavioural aspects concerning the adoption of LCET and sustainable energy use are equally important. Well-designed carbon pricing, both explicit and implicit, should be an important element of the green growth climate policy mix.

Popular science summary

Each of the years 2014 to 2018, the period during which this thesis was written, are among the five hottest years on record. Climate change is happening. But technology progress is also taking place. LED lights have become the standard, dozens of electric vehicle models are on offer and renewable energy outcompetes fossil fuels in many places. Despite this progress, CO_2 emissions have *increased* from 2015, the year when the landmark Paris Agreement was negotiated, to an all-time high in 2018. The target of the Paris Agreement, to limit global warming to well below 2°C by the end of the century, requires all countries to reduce their CO_2 emissions to close to zero by mid-century. Currently, the implementation of policies and measures to get on a pathway towards this target is insufficient.

In the context of climate urgency and technology progress, this thesis investigated green growth climate mitigation policies considering both technology and behavioural change. Green growth policies aim to achieve both GDP growth and CO_2 emissions reductions with economic policy interventions that mainly target the development and adoption of low-carbon energy technologies (LCET). Green growth policies became particularly popular after the 2008/09 global financial crisis, when several countries introduced green fiscal stimulus packages to revitalise their economies. In addition to stimulus spending, green growth policies include subsidies for the development of new LCET and the pricing of CO_2 emissions with taxes or emissions trading schemes. While popular among policymakers, there has been scepticism whether green growth climate policies can achieve what they promise and bring down emissions while economies keep growing.

While the focus of green growth is on technology change, this thesis also explores *behavioural factors and mechanisms* that can affect the design and choice of green growth climate policies. Such behavioural insights build on the rich knowledge from behavioural sciences, and behavioural economics in particular, about how people make decisions. In the past, the idea that people are rational maximisers of their own utility dominated economic policy assessments. However, there has been growing evidence that people are limited in their rationality, their selfishness and their will-power; and in several cases it is found that the deviations from the economic decision-making ideal followed patterns. There is, for instance, evidence that people are short-sighted and largely disregard future costs or benefits, or that they take short-cuts (and do not optimise their decisions) in complex situations. Insights about behavioural factors and mechanisms can support policy-making. Going beyond technology change, they are a promising concept to drive the effectiveness of green growth climate mitigation policies.

Against this background, the objective of this doctoral thesis was to assess the performance of green growth climate policies by going beyond short-term cost-effectiveness and taking into consideration a more behaviourally realistic model of decision-making. This objective was motivated by the uncertainty and lack of empirical knowledge about the effects and the potential of green growth climate policies. Addressing this knowledge gap, the thesis sought to answer the following questions:

- To what extent has green growth been a suitable policy strategy for short to mid-term decarbonisation of the economy?
- What are the main objectives behind green growth policies targeting LCET, and to what extent is climate change mitigation reflected in these objectives?
- What are behavioural factors and mechanisms that drive the public acceptance of different carbon pricing policies?
- What are behavioural factors and mechanisms that drive individuals' adoption of LCET and sustainable energy use practices in response to carbon pricing?
- And finally, how can behavioural insights be integrated into the assessment of green growth policies and what are related policy implications?

To that end, the thesis deployed various research approaches and built on four case studies: the Green Growth Strategy of the Republic of Korea; public financing for low-carbon technology development in the Nordic countries; carbon pricing and regulation targeting energy efficiency of electric appliances in the UK; and different carbon pricing mechanisms tackling emissions in the field of personal transport in Sweden.

The analysis of Korea's Green Growth Strategy between 2009 and 2016 suggests that Korea was not successful in achieving stable CO_2 emissions, let alone CO_2 emissions reductions. In contrast, and despite the Green Growth Strategy, overall CO_2 emissions from fossil fuel combustion grew by 100Mt, which is roughly the total CO_2 emissions of Denmark, Norway and Sweden combined. The strong increase in emissions was mainly driven by economic activity (GDP growth) and could not be offset by improving the energy intensity of the economy or by cleaning up the energy mix. These results question to what extent green growth can be a suitable policy strategy for short to mid-term climate change mitigation, particularly considering the strong connection between economic activity and CO_2 emissions. Results reveal that the problems with green growth policies seems to start already with their objectives, which are not limited to climate change mitigation and may even include seemingly detrimental objectives, which included in the case of Korea the promotion of 'new growth engines' and energy security (via exploration of fossil fuel deposits).

Green growth policy objectives were also explored in the case study on public financing for low-carbon energy technology (LCET) development in the Nordic

Countries. The study investigated several public financing instruments targeting research, development and demonstration of LCET and focused on indicators used to monitor the success of such instruments. The analysis revealed three major trends in the choice of indicators, which reflected underlying policy objectives. First, a strong emphasis on short-term economic performance and return on investment is discovered. Second, it is often highlighted that public financing needs to be additional to (and not replacing) private financing. Third, it is found that performance indicators largely disregarded decarbonisation potential. A growth-oriented innovation policy that provides limited public funding and largely ignores decarbonisation potential might only trigger minor absolute emissions reductions.

In the green growth climate policy narrative, economic instruments play an important role. To achieve larger reductions of CO_2 emissions, one policy that has often been suggested is carbon pricing. Carbon taxes and emission trading schemes increase the cost of emitting CO_2 and thereby (partly) account for the costs associated with climate change, a process that is called internalisation of external costs. By pricing carbon, a price incentive is created to invest in LCET and consume low-carbon goods and services, thereby growing these sectors and creating green growth. The effectiveness and efficiency of this carbon pricing mechanism relies, among various factors, on the economic rationality of market actors, which –as elaborated above – is not always given.

In this context, the case study of the UK market for energy-efficient appliances compared carbon pricing and energy efficiency regulation based on a life-cycle cost model. The aims of the comparison were to see if regulation can capture the external cost of climate change (as carbon pricing does), and to show how stringent carbon pricing needs to be to mimic the effect of efficiency regulation. In the EU, energy efficiency of appliances is mainly regulated through minimum energy performances standards, which set the minimum level of efficiency an appliance must have to be offered on the market. A central objective of these standards is to minimise life-cycle costs, which consist of the appliance price (typically high for very efficient appliances) and the costs of electricity over the expected lifetime (low for efficient appliances). For many appliance types (e.g. tumble dryers and dishwashers) the lowest life-cycle costs can be found somewhere in between the most efficient (and very expensive) appliances and the very inefficient (and costly to run) models.

In this study, a carbon price (reflecting the external cost of climate change) was factored into the electricity price when calculating life-cycle costs. This shifted the life-cycle cost optimum towards more efficient appliances, which implied that standards have to be tightened in order to account for the climate externality. The study also showed that if minimum energy performance standards are tightened by a full efficiency class (e.g. from A+ to A++ labelled dishwashers), carbon prices would have to amount to several hundred EUR per tonne of CO_2 to incentivise a comparable shift. In comparison, additional costs are not very visible in energy performance regulation so that it might have higher political feasibility than explicit carbon pricing and, hence, drive energy efficiency improvements more quickly.

The comparison between regulation and carbon pricing in the context of technology change also had some behavioural implications. The purchase of energy-efficient appliances is known to be affected by bounded rationality and short-sightedness, which implies that future electricity costs (with or without a carbon price) are largely disregarded in the decision process. Hence, short-sighted purchase decisions remain largely unaffected by increasing carbon prices, as only a fraction of the increase is actually considered. This effect could be replicated in the UK modelling study. Regulation, on the other hand, overcomes this challenge. It mandates the minimum efficiency level that is in the self-interest of the average consumer, or the efficiency level that is good for society if the climate externality is accounted for.

To further address behavioural aspects in the context of green growth and carbon pricing, a case study focusing on personal transport in Sweden was conducted. In the study, participants were asked whether and how much they were willing to pay for a climate surcharge on air tickets, for a similar surcharge on fuels and finally for offsetting their CO_2 emissions voluntarily. It was found that most people were willing to pay for the climate surcharge on air tickets, followed by the surcharge on fuels and voluntary offsetting. The acceptance of carbon pricing was partly driven by the mandatory nature of the intervention (voluntary offsetting not being well-accepted). But also the energy use domain (flying versus road transport) and the respective policy context likely played a role. Moreover, respondents' willingness to pay for one tonne of CO_2 emissions was higher for short than for long-distance flights, suggesting that tax rates should be higher in low-cost contexts than in high-cost situations. In addition, women, respondents with a high income, with a left political view, and respondents who were not frequent flyers were all more likely to accept air ticket taxation. Despite this high acceptance, only few respondents felt responsible for reducing their emissions, indicating a gap between attitude and behaviour. Finally, a strong preference for using carbon pricing revenues for climate-related purposes was found, contradicting the common practice of using revenues for the general budget.

When bringing together the research on green growth and behavioural insights, several findings were made. Insights about the behavioural factors and mechanisms that influence technology choices and energy use support the finding that technology change as such is not sufficient for climate change mitigation. Moreover, it was found that behavioural insights can be used to enrich green growth policies addressing the climate externality by taking into consideration behavioural mechanisms, cognitive, motivational, contextual and socio-demographic factors.

While further research is needed to strengthen the results of this thesis, several preliminary implications for policymaking unfold. At the level of policy objectives, green growth policies should have a clear, stringent and irrevocable focus on climate change mitigation if they are to effectively address the targets set by the Paris Agreement. Trade-offs with economic growth must be confronted. At the level of individual green growth climate policies, support to LCET innovation should be extensive and focus on technologies with high decarbonisation potential. In addition, if green growth policies are to address the challenges imposed by the Paris Agreement, then carbon pricing needs to be a central element in the policy mix. Here, the design of carbon pricing matters. Behavioural insights imply that nudging people to offset their emissions voluntarily is not sufficient. Carbon pricing should be mandatory, comprehensive, create a strong price incentive, consider differentiated rates, be framed well, and earmark revenues for climate change mitigation. It should also be considered to target specific groups, reflecting the heterogeneity in the behavioural response to policies. In other sectors, where carbon price signals are not seen or considered, regulations and strict standards are required.

Findings confirm that a more integrated policy evaluation approach is needed to support the design and implementation of green growth climate policies. The fact that many people do not respond with economic rationality to green growth climate policies, such as carbon pricing, implies that policymakers should avoid a simplistic focus on technology change and economic efficiency. Instead, they should put relevance and effectiveness first and maximise the behavioural leverage of climate mitigation policies. Limiting global warming to well below 2°C until the end of the century is a very ambitious target, which needs to be reflected in green growth climate policy.

List of figures

List of papers

Paper 1

Sonnenschein, J., & Mundaca, L. (2016). Decarbonization under Green Growth Strategies? The Case of South Korea. *Journal of Cleaner Production*, *123*, 180–93.

J.S. and L.M. designed and conceptualised the study. J.S. conducted the decomposition analysis, L.M. the econometric analysis. J.S. analysed the results. Both J.S. and L.M. contributed to the manuscript. L.M. supervised the project.

Paper 2

Sonnenschein, J. (2017). Understanding Indicator Choice for the Assessment of RD&D Financing of Low-Carbon Energy Technologies. In D. Arent, C. Arndt, M. Miller, F. Tarp, & O. Zinaman (Eds.), *The Political Economy of Clean Energy Transitions*. Oxford: Oxford University Press.

Paper 3

Sonnenschein, J., Van Buskirk, R., Richter, J. L., & Dalhammar, C. (2019). Minimum energy performance standards for the 1.5°C target: an effective complement to carbon pricing. *Energy Efficiency*, *12*(2), 387–402.

C.D., J.L.R. and R.V.B. had the initial idea and all authors contributed to the research design. Data and LCC optimisation model were provided by R.V.B., and R.V.B, J.S. and J.L.R. contributed to the modelling. Results were analysed by J.S. and J.L.R. Discussion and policy implications were prepared by J.S., J.L.R. and C.D. J.S. drafted the manuscript to which all authors contributed.

Paper 4

Sonnenschein, J., & Mundaca, L. (2019). Is one carbon price enough? Assessing the effects of payment vehicle choice on willingness to pay in Sweden. *Energy Research & Social Science*, *52*, 30–40.

J.S. and L.M. designed and conceptualised the study. J.S. collected and analysed the data. J.S. prepared the discussion and policy implications. L.M. supported the discussion of results and commented on the manuscript.

Paper 5

Sonnenschein, J., & Smedby, N. (2018). Designing air ticket taxes for climate change mitigation: insights from a Swedish valuation study. *Climate Policy*, *0*(0), 1–13.

J.S. designed and conceptualised the study, collected and analysed the data and discussed the results. N.S. supported survey design and data collection, contributed to the discussion of results and commented on the manuscript.

Other publications

Journal Article

Mundaca, L., Sonnenschein, J., Steg, L., Höhne, N. & Ürge-Vorsatz, D. (2019). The global expansion of climate mitigation policy interventions: taking stock, looking ahead. *Environmental Research Communications*. (under review)

Book chapter

Mundaca, L., Samahita, M., Sonnenschein, J. & Seidl, R. (2019). Behavioural economics for energy and climate change policies and the transition to a sustainable energy use – A Scandinavian perspective. In M.Lopes, K. Janda & C. Henggeler (Eds.), *Energy and Behavior: Towards a Low Carbon Future*. Amsterdam: Elsevier. (accepted for publication)

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Sonnenschein, J. (2016). Conditions for the Cost Effective Combination of Emissions Trading and Renewable Energy Support Policies. *Energy Procedia*, *88*, 133–138. https://doi.org/10.1016/j.egypro.2016.06.036.

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Abbreviations

BECCS	bioenergy carbon capture and storage
CDR	carbon dioxide removal
CCS	carbon capture and storage
CO_2	carbon dioxide
CV	contingent valuation
EU ETS	EU Emissions Trading System
GDP	gross domestic product
GGKP	Green Growth Knowledge Platform
GHG	greenhouse gas
IEA	International Energy Agency
IPCC	Intergovernmental Panel on Climate Change
LCC	life-cycle cost
LCET	low-carbon energy technology
LMDI	Logarithmic Mean Divisia Index
MAC	marginal abatement cost
MEPS	minimum energy performance standard
NDC	nationally determined contribution
OECD	Organisation for Economic Co-operation and Development
PV	payment vehicle
RES-E	electricity from renewable energy sources
RCT	randomised controlled trial
RD&D	research, development and demonstration
ROI	return on investment
RQ	research question
SCC	social cost of carbon
SR1.5	(IPCC's) Special Report on 1.5°C
TFC	total final consumption
TPES	total primary energy supply
UNEP	United Nations Environment Programme
VC	venture capital
VCO	voluntary carbon offsetting
WTP	willingness to pay

1. Introduction

> The human race is challenged more than ever before to demonstrate our mastery – not over nature but of ourselves. – *Rachel Carson*

Climate change, global warming, climate breakdown! Irrespective of the label, anthropogenic greenhouse gas (GHG) emissions have begun to drastically alter our life on earth (Hoegh-Guldberg et al., 2018). Impacts include among others the rapid melting of glaciers and ice sheets, the melting of permafrost, sea level rise, the spread of extreme heat, the change of precipitation patterns, shifting of plant and animal ranges as well as coral die-off (Hoegh-Guldberg et al., 2018; IPCC, 2014). So far we have failed to reduce global GHG emissions, failed to introduce sufficiently effective climate change mitigation policies, and even failed to truly understand the mitigation challenge – let alone the failure to secure better opportunities for future generations. In contrast, climate change is even intensifying as global carbon dioxide (CO_2) emissions are still increasing and so is the global average temperature (Allen et al., 2018). Ever increasing emissions and the rise of global temperature are not consistent with the 'planetary boundaries'[1] to our life on this planet (Steffen et al., 2015) and challenge the functioning of the economy system in itself (Grubb, Hourcade, & Neuhoff, 2013).

For global warming, the planetary boundary has – in a politico-scientific process – been defined in the targets of the Paris Agreement (Paris Targets)[2], which aim at:

> Holding the increase in the global average temperature to well below 2°C above pre-industrial levels and to pursue efforts to limit the temperature increase to 1.5°C above pre-industrial levels, recognizing that this would significantly reduce the risks and impacts of climate change. (United Nations, 2015)

Several scenarios were modelled to show how the Paris Targets could be reached. Most scenarios require the complete decarbonisation of the energy system by mid-

[1] The idea of natural boundaries to human activities is not new (cf. Club of Rome, 1972) but shifted from the idea of limited resources to the idea that going beyond certain thresholds and tipping points might cause rapid and irreversible environmental change.

[2] It is important to note that this boundary is the outcome of political negotiations and as such subject to uncertainties (Peters, 2016); achieving it does not necessarily mean that the worst impacts of climate change can be avoided (Allen et al., 2018).

century and, thereafter, the deployment of carbon dioxide removal (CDR) options (Rogelj, Shindell, et al., 2018). The rate of emissions reductions has to be higher than previously achieved in any country in the world; and most of the CDR options are still to be developed, tested and implemented at scale (Field & Mach, 2017). While limiting global warming to well below 2°C is already an unprecedented challenge, there are severe doubts whether it is feasible at all, even under very optimistic assumptions, to limit warming further to 1.5°C (Rogelj, Popp, et al., 2018).

While scenarios clearly illustrate the required speed and scale of emissions reductions, climate policy to achieve emission reductions remains vastly insufficient (Geden, 2016; IPCC, 2018; UNEP, 2018). To begin with, nearly 90% of global GHG emissions were covered by economy-wide GHG reduction targets in 2017 (Iacobuta, Dubash, Upadhyaya, Deribe, & Höhne, 2018). However, this coverage does not tell much about the ambition of the respective reduction targets. Accordingly, it has been found that the national targets expressed in countries' nationally determined contributions (NDC) under the Paris Agreement are inconsistent with the Paris Targets (UNEP, 2018). Fully implementing the NDCs would result in a temperature increase of about 3°C (Rogelj, Shindell, et al., 2018).

Going beyond targets and looking at actual policy interventions, the situation is even more challenging. By 2017 about 70% of global emissions were covered by national legislation and climate strategies (Iacobuta et al., 2018). This value may sound substantial to start with, but it does not tell anything about the timing, type and stringency of the respective legislation. In fact, it is shown that climate mitigation policies are insufficient in virtually every sector, including the extraction of fossil fuels (Erickson, Lazarus, & Piggot, 2018; Piggot, Erickson, Asselt, & Lazarus, 2018), the regulation and pricing of emissions from fossil fuel combustion (de Coninck et al., 2018; Mundaca & Markandya, 2016; World Bank & Ecofys, 2018), as well as energy demand side policies (Mundaca, Ürge-Vorsatz, & Wilson, 2018). While there is much agreement on the general inadequacy of current climate mitigation policy, there is debate about what the right policies are for reaching the transformative Paris Targets (de Coninck et al., 2018).

1.1. Green growth and decarbonisation

One strong discourse that influenced climate and energy policies in recent years was 'green growth' (conceptual details in Section 3.2) (Bäckstrand & Lövbrand, 2016). This policy discourse encompasses the concurrent achievement of economic growth and stable or lower GHG emissions via de-coupling (Bowen & Hepburn, 2014; IEA, 2009; OECD, 2011b). After the 2008/09 Financial Crisis it was even argued that 'a Global Green New Deal, if implemented effectively and swiftly, has the potential to revive the world economy and reduce its vulnerability to […] climate-induced risks.'(Barbier, 2010a, p. 20). However, the concept of green growth has also been questioned as so far there is not much empirical evidence that the absolute de-coupling of emissions from economic growth has worked. As shown in Figure 1, large improvements have been made in improving the carbon intensity of economic activity, whereas absolute CO_2 emissions increased consistently, with only few exceptions that coincided with global crises.

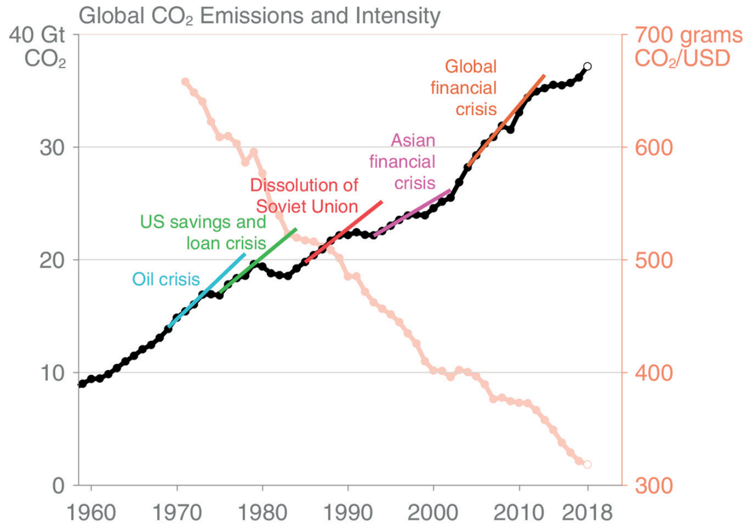

Figure 1
Global CO_2 emissions and carbon intensity of the economy 1960-2018 (source: Global Carbon Project).

Besides fiscal stimulus, which aims at avoiding emissions rebounds after such economic crises, green growth policies mainly comprise economic policy instruments, whereas regulatory instruments do not have a prominent role and their economic efficiency has been questioned (OECD, 2013). Therefore, it is not surprising that reviews of policy instruments have found that economic instruments were at the centre of climate change mitigation and energy policy so far, above all government subsidies and carbon pricing interventions (Mundaca & Markandya, 2016; Mundaca, Sonnenschein, Steg, Höhne, & Ürge-Vorsatz, under review).

This thesis analysed green growth climate change mitigation policies with a focus on economic policy instruments. The specific policies that were within the scope of the analysis are green fiscal stimulus, public subsidies for research, development and demonstration (RD&D) of low-carbon energy technologies, minimum energy performance standards (MEPS) for electric appliances and carbon pricing.

Among economic policy instruments, carbon pricing stands out in the green growth discourse to address climate change (Barbier, 2010a; Bowen, 2015; Jouvet & de

Perthuis, 2013; Reilly, 2012). Explicit carbon pricing instruments include carbon taxes and emissions trading schemes. Many economists have argued that these carbon pricing mechanisms can, at least in theory, achieve emission reductions in a cost-effective way, as the emissions abatement technologies with the lowest marginal costs will be implemented first (Aldy & Stavins, 2012; Goulder & Schein, 2013; Nordhaus, 2007; Rogelj, Shindell, et al., 2018; Somanathan et al., 2014; N. H. Stern, 2006; Weitzman, 2014). The Special Report on 1.5°C of the IPCC (SR1.5) concludes, after assessing scenarios from the Integrated Assessment Modelling literature, that 'explicit carbon pricing is relevant but needs to be complemented with other policies to drive the required changes' (Rogelj, Shindell, et al., 2018, p. 153). This call for a policy mix already hints at due scepticism towards the effectiveness of both stand-alone carbon pricing and the green growth concept with its focus on economic policy interventions.

Market failures on the markets for low-carbon energy technologies (LCET) have been at the centre of the critical debate around carbon pricing so far, and it has been argued that additional policy interventions are needed (de Coninck et al., 2018; Houde & Spurlock, 2016; Jaffe, Newell, & Stavins, 2005; Lehmann & Gawel, 2013). Complementary energy efficiency policy may even have the potential to increase the cost-effectiveness of carbon pricing (Brown & Li, 2019). In addition to *market* failures, however, there is increasing evidence that also *behavioural* aspects affect the response to carbon pricing interventions (Gillingham & Palmer, 2014; Gowdy, 2008; Lunn, 2015).

Considering both the need for rapid decarbonisation and the scepticism about green growth, this thesis aims to improve the understanding how (much) green growth policies, and carbon pricing in particular, can contribute towards reaching the Paris Targets. To that end, the thesis work embarks on an economic policy analysis that goes beyond short-term cost-effectiveness and takes into consideration both the actual track-record of green growth policies until now and a more behaviourally realistic model of decision-making in response to carbon pricing.

The following section will highlight current knowledge gaps in the assessment of green growth policies and the analysis of behavioural aspects of carbon pricing (Section 1.2). This is followed by the main objectives and research questions of this thesis, including further justifications (Section 1.3), and its scope (Section 1.4).

1.2. Knowledge gaps

Currently, it is difficult to judge the contribution of green growth climate policies to achieving the Paris Targets and to assess the potential of behavioural insights in this regard. This is largely driven by the lack of empirical assessment of related policies.

At a general level it has been argued that 'society is hampered in using natural science knowledge of climate change because of gaps in the knowledge of economic and social dimensions of climate change' (Burke et al., 2016, p. 293). Similar arguments have been made about the deficient use of social science-related disciplines in energy and climate research (Sovacool, 2014; P. Stern, Sovacool, & Dietz, 2016), leading to the conclusion that 'the climate change community, and related policymakers need to recognize that energy production, consumption, and policy are both social and technical domains' (Sovacool et al., 2015, p. 95). The social domain goes beyond economic analysis, which has been argued to be the dominating discipline in the scientific review and assessment work carried out for the IPCC (Victor, 2015). However, even within climate change economics, much more progress is needed with respect to the economic assessment of climate policies (Burke et al., 2016).

When zooming in to the field of green growth policies for climate change mitigation, there have been only few quantitative analyses of dedicated green growth policies and there is a clear lack of robust empirical data to support or challenge the win-win promise of the green growth concept (Bowen & Hepburn, 2014; Mundaca, Neij, Markandya, Hennicke, & Yan, 2016). More specifically, macroeconomic energy-economy and climate indicators are frequently monitored and reported (OECD, 2017b), but they are rarely discussed in the context of country-specific policies. Furthermore, it is unclear to what extent countries pursue an environmental agenda with their spending on LCET innovation. The lack of research on the performance of green growth policies may help to explain the discrepancy between the political optimism about green growth policies and academic scepticism of green growth's environmental effectiveness.

In the case of behavioural insights, first applications to climate mitigation policy have been attempted, but much research, both at the conceptual and at the empirical levels, still needs to be carried out. This observation is in line with the 5th IPCC Assessment Report, which found that 'more research that incorporates behavioural economics into climate change mitigation is needed. For instance, more work on understanding how individuals and their social preferences respond to (ambitious) policy instruments and make decisions relevant to climate change is critical' (Kolstad et al., 2014, p. 258). In 2016 a special issue in *Applied Energy* solely dedicated to the assessment of green energy economy policies and measures also identified a lack of knowledge regarding the role and performance of behavioural-oriented policies in the green growth discourse (Mundaca et al., 2016).

This knowledge gap seems to persist as the latest IPCC research synthesis in the SR1.5 found a 'limited understanding and treatment of behavioural change and the potential effects of related policies in ambitious mitigation pathways' (de Coninck et al., 2018, p. 391). While SR1.5 identified evidence about climate or energy policies targeting market failures, no study addressing policy strategies based on behavioural insights in the 1.5°C context was found (de Coninck et al., 2018). In a similar vein it has been argued that 'studies are urgently needed to systematically examine how to achieve the accelerated and wide-scale changes in behaviour, organizations, institutions and political systems that are assumed in scenarios and pathways' (Steg, 2018, p. 761). A review of psychology research on individual behaviour that drives climate change identified future research needs in particular for 'non-financial factors influencing high-impact household behaviours that could mitigate climate change, particularly the adoption of environmentally friendly technologies, and the ways in which these factors interact with monetary inducements' (Clayton et al., 2015, p. 642).

Finally, the integration of behavioural insights into the overarching green growth policy discourse has not advanced very far. In order to investigate how green growth policies can be improved based on behavioural insights, the Green Growth Knowledge Platform[3] (GGKP) set up an international 'Behavioural Insights Working Group', which was active between 2015 and 2017 and released a working paper on the topic (Castro de Hallgren & Root-Bernstein, 2018). While the paper presents a conceptual background to behavioural interventions and a valuable collection of case studies, it does not highlight how behavioural insights can be used in the specific context of green growth policies (as opposed to general environmental or climate policy).

1.3. Research objectives and questions

The overarching aim of this PhD thesis was to contribute to a better understanding and design of green growth climate mitigation policies by conducting and improving (economic) policy assessment. In order to achieve this aim, and considering the research gaps outlined above, the research was guided and scoped by two main objectives combining top-down and bottom-up research approaches.

[3] This initiative was set up in 2012 by the World Bank, UNEP, OECD and the Global Green Growth Institute to address major knowledge gaps in green growth theory and practice.

Objective 1: Improve knowledge of how green growth policies affect CO_2 emissions and of their potential contribution for reaching the Paris Targets

This first objective was motivated by the above-mentioned gap in evaluations of green growth policies. More knowledge was needed to assess whether following a green growth policy strategy is a viable way to address the climate externality and make a significant contribution towards achieving the Paris Targets. Under this objective, specific research questions were asked in order to scope down the research and come to tangible conceptual contributions and policy implications.

RQ 1: To what extent has green growth been a suitable policy strategy for short to mid-term decarbonisation of the economy (as expressed in key macroeconomic drivers)?

To address this RQ, the Republic of Korea (hereinafter 'Korea') was taken as case study (Paper 1) [4]. At the time the research was carried out, Korea was praised as the world leader in green growth policy and economic efforts (Barbier, 2010b). Due to its international relevance, volume and focus on climate change mitigation, the Korean Green Growth Strategy was chosen as a suitable case to advance the knowledge on the potential of green growth policies to mitigate climate change.

In addition to the empirical assessment of quantitative outcomes, knowledge was also needed about the interplay between green growth and climate policy. It is unclear to what extent climate change mitigation (towards reaching the Paris Targets) was reflected in green growth policy objectives and design. In this context RQ 2 covered green growth policies targeting LCET and their objectives.

RQ 2: What are the main objectives behind green growth policies targeting LCET, and to what extent is climate change mitigation reflected in these objectives (and in the resulting policy design)?

To address this question, the Nordic countries and their support for research, development and demonstration (RD&D) activities in the LCET sector were taken as case study (Paper 2). Building upon the findings from Paper 1, the Nordics were chosen as a case because they had a well-established LCET sector (Irandoust, 2016) and several support policies in place. In order to achieve the Paris Targets, RD&D for LCET innovation needs to be scaled up (de Coninck et al., 2018; Rogelj et al., 2015), which in turn requires public support with financing. In this context it is critical to know what motivates the public sector to support the financing of RD&D of LCET. It is unclear to what extent decarbonisation potential plays a role in the assessment of public RD&D financing in particular and green growth policies in general.

[4] See Table 2 in Section 2.2 for an overview of the research papers included in this thesis.

Moving on from the top-down perspective of national green growth policies, the second objective of this research takes a bottom-up perspective and deals with individual decision-making in the context of carbon pricing – a key policy that has been put forward by the green growth policy discourse.

Objective 2: Provide a better understanding of how behavioural mechanisms influence the outcomes of carbon pricing.

As outlined above, knowledge gaps have been identified regarding the behavioural mechanisms driving climate change mitigation in response to policy interventions. Within this field, the effect of carbon pricing on behaviour is particularly complex and goes beyond providing a simple economic incentive to adopt LCET (Bolderdijk & Steg, 2015; Klenert et al., 2018). To start with, carbon pricing can only take effect if it is implemented, which depends among others on its acceptance (Klenert et al., 2018). There is some evidence that the acceptance of carbon pricing policies depends both on economic factors, such as the price level, and on various behavioural factors (Bristow, Wardman, Zanni, & Chintakayala, 2010), but more research is needed on the factors that drive acceptance. If such behavioural factors are known, they can be considered to inform policymaking. Behavioural factors and mechanisms behind acceptance were captured in RQ 3.

RQ 3: What are the behavioural factors and mechanisms that drive the public acceptance of different carbon pricing policies?

RQ3 was addressed in a survey study that investigated people's acceptance levels of different carbon pricing mechanisms, their behavioural determinants and contextual factors (Papers 4 & 5). Acceptance can be researched both for a carbon pricing mechanism as such and for different carbon price levels. This was addressed in the survey study by researching both people's willingness to pay *in principle* for CO_2 emissions under different pricing schemes, and by investigating the *amount* people were willing to pay in the respective cases.

For the effectiveness of carbon pricing it is, however, not only relevant that an ambitious carbon pricing scheme is accepted and put in place to start with, but that it results in the adoption of LCET and sustainable energy use. The influence of behavioural factors and mechanisms on the effectiveness of carbon pricing was captured in RQ 4.

RQ 4: What are behavioural factors and mechanisms that drive individuals' adoption of LCET and sustainable energy use in response to carbon pricing?

This RQ was partly addressed by the above-mentioned survey study (in particular Paper 5). Moreover, it was addressed in a modelling study that dealt with the market for energy efficient appliances and compared two policy approaches to promote more efficient appliances, namely carbon pricing and energy performance regulation, and their behavioural implications (Paper 3).

The research questions 1–4 deal separately with the assessment of green growth policies and behavioural insights regarding carbon pricing. Green growth policies are, however, not limited to carbon pricing; and behavioural insights for policy choice and design can be gained and applied in any policy domain. Therefore, there might be cross-cutting issues of climate policy assessment that relate to both green growth and behavioural insights. This is addressed in RQ 5, which deals with the implications of researching green growth and behavioural insights for both policy analysis and policymaking.

RQ 5: How can behavioural insights be integrated in the assessment of green growth policies and what are related policy implications?

This RQ was not addressed by one specific paper or study but by synthesising the thesis' research findings and integrating the recent literature. This synthesis includes both a discussion of conceptual issues, that is, how to account for behavioural insights in the assessment of green growth policies, and a discussion of key findings with implications for climate mitigation policy.

1.4. Scope

The two research objectives and five research questions already de-limit the scope of this thesis. In addition, the scope of the research was narrowed down with respect to green growth climate mitigation policies, including fiscal stimulus, RD&D for LCET, MEPS and carbon pricing (see Table 1). However, it needs to be acknowledged that the research still dealt with a broad range of topics, which made it impossible to meet the above objectives in a comprehensive way. Therefore, and in addition to this scoping section, needs for future research that could not be covered in this thesis are outlined in Section 6.4.

The thesis had a clear focus on industrialised countries. There are certainly large research gaps in the analysis of climate policy and green growth in developing countries, too (Burke et al., 2016; de Coninck et al., 2018; Mundaca et al., 2016). However, industrialised countries caused most of the past GHG emissions and are still the largest emitters, they have the largest resources to address climate change, and, to a certain extent, have a track record of climate mitigation policy. Besides, the focus on industrialised countries is motivated by pragmatic arguments, such as research funding and data availability.

Table 1
Outline of the research scope in different domains.

Domain	Within scope	Outside the scope
Policy	Green growth climate mitigation policies	Other climate mitigation policies, adaptation, geo-engineering
Policy Analysis	Policies and measures	Policy processes, politics
Geography	Industrialised countries	Developing countries
Actors	Governments, individuals	Organisations
Climate change	Anthropogenic CO_2 emissions	Other GHGs, natural factors and carbon sinks

Further, it is important to note that decarbonisation via green growth climate *mitigation* policy is only one of the options to deal with the challenge of climate change. Further options include adaptation and geo-engineering (de Coninck et al., 2018), but these were outside the scope of this thesis. Moreover, this thesis focused on policies addressing anthropogenic CO_2 emissions, which are only a part of all GHG emissions and radiative forcing. They are, however, the dominating driver of global warming (Rogelj, Shindell, et al., 2018).

The focus of policy assessment was clearly on climate *policies*, that is the combination of policy instruments and measures (see Section 2.4). Issues concerning the policy *process*, the *politics* of climate change, and climate *ethics* were outside the scope. While public acceptance was included in the research, political acceptability was not.

The thesis also had a clear methodological focus, which is quantitative economic policy assessment. Economic analysis is, however, very broadly understood here and includes 'injections' from policy evaluation theory (Papers 1 & 2), innovation systems theory (Paper 2) and psychology (Papers 4 & 5).

1.5. Target audience

Policy assessment is ideally conducted to generate results that can be utilised in the policy-making process (Patton, 2008). Besides the actual findings of a policy assessment, several aspects are important to take into consideration when using such results, including among others the aspects of the policy intervention that were considered and the intended user (Alkin & King, 2017). While the ultimate goal of the policy assessments conducted for this thesis was that they are used in the selection and design of actual policies, the more likely (and immediate) way results are used is of a more conceptual nature, e.g. by feeding into the design of future policy assessments or by providing knowledge for future research. Accordingly, the targeted users of this thesis' findings were not only policymakers directly, but academics, policy think tanks and experts in relevant public institutions (e.g. Energy Agency, Ministry for Environment).

1.6. Thesis outline

The thesis is presented in six chapters and one appendix that contains the five research papers that were published in the context of this PhD project. After the introduction in Chapter 1, Chapter 2 introduces the general research approach of this thesis, gives some background about policy assessment and presents the unit of analysis (green growth climate mitigation policies). Moreover, the specific methods for data analysis and data collection that were used in the research papers are summarised.

Chapter 3 provides the conceptual framing of the thesis. The main concepts covered are the (climate change mitigation) policy process, green growth and behavioural insights. These three concepts are brought together in a combined framework for policy assessment, which is presented at the end of the chapter.

Chapter 4 summarises the five research papers. These summaries include the background of the respective study, its connection to the conceptual framework, key results and their relation to the research questions, and policy implications.

Chapter 5 then synthesises and discusses the results of the papers horizontally, taking into consideration findings from literature. The discussion is organised along the five research questions. It also includes reflections on the strengths and limitations of the conceptual framework and the methodological approach.

Chapter 6 concludes the thesis by summarising its main findings. In addition, the main contributions to theory are highlighted, overarching policy recommendations are listed, and areas for future research are outlined.

2. Research design and methods

This chapter goes from the general to the specific. It starts by introducing the fundamental positioning of this thesis' research (Section 2.1) and goes on to outline the overarching research approach (Section 2.2). This outline briefly introduces the research papers that are included in this thesis and puts them into the context of the research objectives. After that, the general policy assessment approach is outlined (Section 2.3) and the unit of analysis is defined and specified (Section 2.4). This is followed by an overview of the specific research methods that were used in the papers of this thesis, including methods for data analysis (Section 2.5) and data collection (Section 2.6). Finally, the approaches that were used to increase the validity of results (Section 2.7) and to warrant transparency (Section 2.8) are described.

2.1. Research positioning

The aim of this thesis, to contribute to a better design and implementation of green growth climate mitigation policies by conducting and improving (economic) policy assessment, required research that is solution-oriented (Watts, 2017), interdisciplinary (Stock & Burton, 2011), and falsifiable (Popper, 1963). Starting from the latter, this thesis took the epistemological position that only falsifiable research results are scientifically sound. This is based on Karl Popper's idea that a good test of a theory is to try to falsify it and that an unfalsifiable (not: unfalsified!) theory is unscientific (Popper, 1963). This epistemological position is reflected in the choice of research methods (see Sections 2.5 and 2.8), which put great emphasis on replicability by making data available and analytical steps explicit.

From an ontological perspective, this thesis does not take one dominating position. While focused on the *economic* analysis of green growth climate mitigation policy, the research presented in this thesis is critical towards mainstream economic theory and methods, but does not dismiss them from start. The usefulness of economic concepts, methods and also policy instruments depends on well-informed judgement and can neither be assumed nor denied a priori (Strunz, Klauer, Ring, & Schiller, 2016). The *interdisciplinary* nature of this thesis lies in its integration of economics, policy evaluation and psychology in the research of climate mitigation policies[5]. This integration took place at the level of literature study, by collaboration and exchange with researchers from different disciplines, and by applying multiple methods.

While grounded in theoretical concepts, the aim of this thesis was also to be relevant for policy-making. Its ambition was to be located in 'Pasteur's quadrant', which describes research that is *use-inspired* and still advances fundamental understanding (Stokes, 1997). Hence, this research aims to be distinct from basic social science research that aims at fundamental (theoretical) understanding, and distinct from purely applied research that builds on theoretical concepts without contributing to their further development. The 'use' that inspired this thesis is the choice, design and mix of green growth climate mitigation policies. *Policy-oriented* research focuses on actionable factors or variables (Hakim, 2000) such as policy interventions, which are the unit of analysis of this thesis (see Section 2.4).

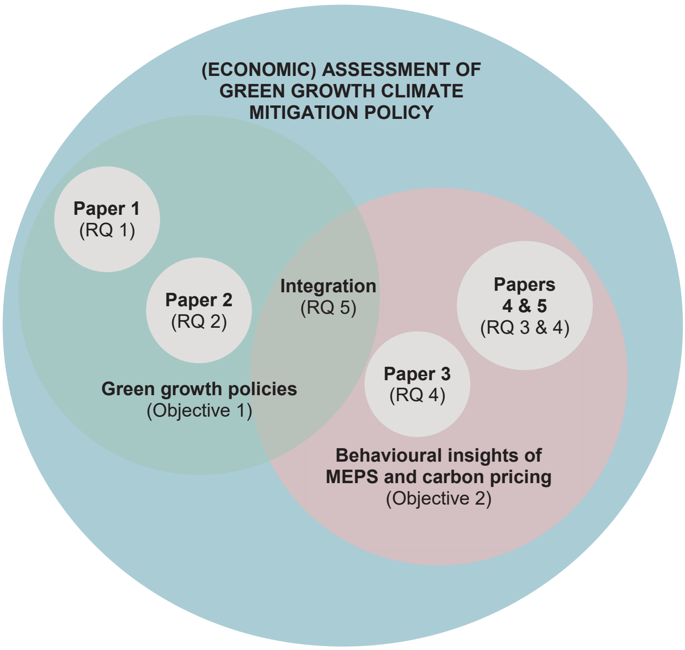

Figure 2
Mapping of research outputs against research objectives and questions.

[5] The complex and 'super-wicked' nature of the climate change challenge (Lazarus, 2008) means that policies are likely to fail if they only consider engineering solutions, if they only consider economic solutions, or if they only build on behavioural change. Economic research of policies in this field requires an interdisciplinary approach (Bhaskar, 2010).

2.2. Overarching research approach

The research for this thesis was carried out by conducting several case studies (see Figure 2)[6]. In the context of policy assessment, case studies allow for the review and testing of hypotheses from academic literature in the specific context of a certain case (Hakim, 2000; Yin, 2014). While individual cases do not allow for generalisations, they provide empirical evidence in favour of or against previously established concepts and findings and thereby contribute to the development of theory. In order to relate the research to such concepts and findings, the case studies were prepared by reviewing relevant literature and their findings were discussed in the context of previous research in the respective area.

The set of papers in this study is very diverse with respect to the geographical locations, the policy interventions that are covered (Section 2.4), the methods for analysis (Section 2.5), and the methods for data collection (Section 2.6). What ties the papers together is that they all deal with the analysis of green growth climate mitigation policies for LCET and sustainable energy use. Moreover, all analyses explicitly take into consideration market failures and/or behavioural insights, which reflects the main objectives of this thesis. An overarching conceptual framework for all research papers is presented in Section 3 below. A synthesis of research findings (addressing RQ 5) is provided in Chapter 5.

Table 2
Overview of research outputs and their main methodologies.

Paper #	1.	2.	3.	4.	5.
Case	South Korea	Nordic countries	UK	Sweden	Sweden
Policy	Green Stimulus	Low carbon RD&D finance	MEPS & carbon pricing	Carbon pricing	Air ticket taxation
Published as	Journal article	Book chapter	Journal article	Journal article	Journal article
Main character	Quantitative	Qualitative	Quantitative	Quantitative	Quantitative
Main data source	Statistical databases	Interviews & document research	Online market-places	Survey	Survey
Method(s) for data analysis	LMDI decomposition & econometrics	Criteria-based indicator assessment	Life-cycle cost modelling & econometrics	Contingent valuation & statistical tests	Contingent valuation & econometrics

[6] In addition to the five research papers that are appended to this thesis, further research outputs include conference papers (Sonnenschein, 2016b, 2016a) and policy briefs (Sonnenschein, 2016c; Sonnenschein, Richter, & Dalhammar, 2018), which are not presented separately but partly considered in the discussion. A full list of publications is included in the front matters.

2.3. Policy assessment

The research papers that form this thesis made use of several different policy assessment methods and focused on specific aspects of the respective policies, rather than evaluating their overall performances. Therefore, the term policy *assessment* was chosen over *analysis* or *evaluation*. This allowed for the use of a variety of different research methods and avoided a narrow theoretical positioning. While the main term used in this thesis is policy assessment, it still covers various aspects of policy evaluation, including the use of a *framework* to conceptualise the policy intervention process that is assessed, the use of different *evaluation criteria* and the *utilisation* of results to derive policy implications.

Policy evaluation was defined by Vedung (2009, p. 3) as 'careful retrospective assessment of the merit, worth and value of administration, output and outcome of government interventions, which is intended to play a role in future, practical action situations'. While this definition confines policy evaluation to ex post analysis, various authors also include ex-ante evaluation in their definitions (F. Fischer, 1995; Mickwitz, 2003). Including a more forward-looking and transition-oriented approach to policy analysis and evaluation appears to be appropriate considering that past climate mitigation policies have not even closely resulted in sufficient emission reductions.

Within policy assessment there is room for manoeuvre to adopt a more forward-looking and transition-oriented approach (or not); and a key determinant of the chosen approach is the selection of *evaluation criteria*. The selection of criteria determines the focus of the analysis and is ultimately a normative choice (Mickwitz, 2003). Throughout the thesis *environmental effectiveness*, that is CO_2 emissions reductions towards reaching the transformative Paris Targets, was the central criterion used in the analysis. This reflects the objective of green growth climate mitigation policies to reach climate targets (besides growing the economy). In addition, effectiveness in terms of *technology change* is another central objective of green growth climate policies (Popp, 2012) and it was addressed by Papers 1, 2 & 3. Finally, also the *relevance* of green growth policy objectives in the context of the Paris Targets was discussed based on Papers 1 & 2[7]. In order to include these evaluation criteria in the assessment of green growth climate mitigation policies, different steps of the policy intervention process were considered, including societal needs, policy objectives, policy inputs, their outputs, outcomes and finally impact (see also Figure 3 in Section 3.1).

[7] Other evaluation criteria that were not at the core of this thesis about green growth climate policy assessment, but were touched upon in the research papers, were *public acceptance* (Papers 4 & 5), *co-benefits* (Paper 2), and *cost-effectiveness* (Paper 3).

2.4. Unit of analysis: Green growth climate mitigation policies

The unit of analysis of this thesis is green growth climate mitigation policies. According to the IPCC, climate mitigation policies include both policy instruments (e.g. economic instruments and regulations) and measures (e.g. adoption of technologies, practices and behavioural change) (Allwood, Bosetti, Dubash, Gómez-Echeverri, & von Stechow, 2014). These elements are in line with another definition of policy interventions, which describes them as 'any regulation, policy, program, measure, activity, or event that aims to influence behaviour' (Wilson & Dowlatabadi, 2007). Such a broad definition was chosen to accommodate for the different types of policies included in this thesis, reaching from large national strategies to small behavioural interventions.

The framing of policy instruments in the field of climate change mitigation (Kolstad et al., 2014) and energy (Jaccard et al., 2012) typically follows the same taxonomy and includes:

- Regulations (incl. bans, mandates, standards),
- Economic instruments (incl. taxes, fees, subsidies, preferential loans),
- Information (incl. advice, promotion, campaigns, guidance), and
- Provision of public goods (incl. infrastructure).

Among these instruments, the focus of the thesis was on green growth climate mitigation policies, which mainly include economic instruments, and to a lesser extent regulations and information. The specific policy instruments and measures that were analysed in this thesis are briefly elaborated below, and more information is contained in the appended research papers.

Green fiscal stimulus

Among the analysed interventions, green fiscal stimulus is the only one that constitutes a larger policy programme (see also Section 3.2). Fiscal stimulus refers to increased government spending (consumption, investments), which is typically applied to stimulate the economy in times of economic crisis. Other things being equal, increased economic activity, the target of fiscal stimulus, leads to increased resource consumption and emissions. The core idea behind *green* fiscal stimulus is to avoid the negative environmental impacts of stimulated economic growth by targeting sectors and technologies with low environmental impacts, e.g. LCET (Barbier & Markandya, 2013).

An analysis of Green Fiscal Stimulus was carried out in Paper 1, which took the Korean Green Growth Strategy as a case study.

Public sector financing of research, development and demonstration

Public sector financing of Research, Development and Demonstration (RD&D) comes in different forms, including grants, preferential loans, guarantees, and even public venture capital (VC) and public equity instruments. Public sector financing is argued to be beneficial for society if it helps to fix market failures that prohibit optimal investments into RD&D by the private sector, including knowledge spill-overs, risk aversion and information asymmetries (Arrow, 1972; Dasgupta & Stiglitz, 1980; Grünfeld, Iverson, & Grimsby, 2011; Jaffe et al., 2005). Arguments that public financing crowds out private investments have been largely dismissed (Ali-Yrkkö, 2004). Furthermore, private investments in energy RD&D tend to 'reinforce the existing energy paradigm while the public sector is focusing on new energy technologies that support wider policy objectives' (A. Rhodes, Skea, & Hannon, 2014, p. 5601). In order to achieve the decarbonisation of industrial economies by mid-century, innovation in LCET is urgently needed and this, in turn, requires targeted policy support (Sandén & Azar, 2005). It has been argued that the experimentation in pilot and demonstration plants is particularly important for the development of LCET and requires public support (Hellsmark, Frishammar, Söderholm, & Ylinenpää, 2016).

Public sector financing of RD&D of LCETs was covered in Paper 2, which explored different support instruments in the Nordic countries and the rationale behind them.

Minimum energy performance standards

Minimum energy performance standards (MEPS) fall into the category of regulations and can be defined as 'legally enforced thresholds for an individual product or group of products, set at a level to exclude a proportion of the worst performing products in the marketplace' (Ellis, 2007, p. 18). The exact definitions of MEPS differ between countries. In Japan, for example, MEPS may be calculated as a fleet average, allowing manufacturers to compensate for products that do not comply with the MEPS by putting other very efficient appliances on the market. In the EU, in contrast, every product has to comply with MEPS (Siderius & Nakagami, 2013). There are different ways to set MEPS, but they typically target consumer benefits, e.g. by reducing the life-cycle costs associated with the use of an appliance. While MEPS are certainly not at the centre of the green growth policy paradigm, they were included in this thesis for two reasons. First, they can be regarded as an *implicit* carbon pricing policy and can be contrasted with explicit carbon pricing (see following section). Second, there are indications from the EU (BSH, Philips, Electrolux, & Camfil, 2012) and the Nordic countries (Jönbrink & Melin, 2008) that MEPS for electric appliances can create a competitive advantage for experienced companies with high technical know-how, and thereby contribute to green growth.

MEPS were covered in Paper 3, which looked at MEPS for electric appliances in the EU, taking the UK market as an example.

Carbon pricing

Carbon pricing instruments[8] fall into the category of economic instruments and come in different forms (Aldy & Stavins, 2012). A key differentiation is between explicit carbon pricing instruments, including carbon taxes and emissions trading schemes, and implicit carbon pricing, which includes various instruments (e.g. energy taxes, standards and regulations) that imply or assume a certain carbon price level, e.g. in underlying cost-benefit analysis (de Coninck et al., 2018). Moreover, large differences exist with respect to the scope of carbon pricing instruments, their ambition level and design features such as exemptions, reductions and refunds (World Bank & Ecofys, 2018).

Several carbon pricing instruments were covered in the papers of this thesis, including a carbon price on electricity in Paper 3 and fuel and air ticket surcharges in Papers 4 & 5. Furthermore, Paper 3 also modelled different implicit carbon prices associated with MEPS for appliances and confronted them with explicit carbon pricing.

Policy instrument mix

This thesis analysed several climate mitigation policies. These policies, however, do not exist in isolation but interact with each other and with further (climate) policies. Interactions of climate-energy policies exist between sectors and between different jurisdictional levels (Goulder & Stavins, 2011). It has been argued that policy overlaps might lead to inefficient outcomes, in particular for renewable energy support policies and energy efficiency regulation (Fankhauser, Hepburn, & Park, 2010; C. Fischer, Torvanger, Shrivastava, Sterner, & Stigson, 2012).

On the other hand, a policy mix may be justified to deal with various market failures that cannot be solved by carbon pricing alone, including knowledge spillovers of innovation, concentrated market power and information asymmetries (Gillingham, Newell, & Palmer, 2009; Houde & Spurlock, 2016; Lehmann & Gawel, 2013). Moreover, the efficiency of policy (mixes) changes if a longer time-perspective is considered (Sandén & Azar, 2005) and a dynamic notion of efficiency is used (del Río González, 2008). If multiple policy objectives (beyond addressing climate change) are considered, a policy mix might not only be justified but required (Lehmann & Gawel, 2013).

[8] See further elaborations on carbon pricing in the section on ‘Economic instruments’ in Section 3.2.

In this thesis, policy mixes and interaction effects are touched upon in Papers 1–3 and in the discussion of this thesis (Chapter 5). Moreover, the topic of interactions between emissions trading and LCET support policies was addressed in a peer-reviewed conference paper written in the context of this thesis (Sonnenschein, 2016a).

Measures

The measures that are driven by above-listed policy instruments are 'technologies, processes or practices that contribute to mitigation' (Allwood et al., 2014). The measures that are covered by this thesis include the development of LCETs (Papers 1 & 2), the adoption of LCET in general (Paper 1) and energy efficient electric appliances in particular (Paper 3), and reductions in transport in general (Papers 4) and air travel in particular (Paper 5).

2.5. Methods for data analysis

Due to the width of cases studies and topics covered and the interdisciplinary research approach, and in order to facilitate wide learning in the PhD research process, different data collection methods were employed. These included both quantitative and qualitative analysis methods. A brief overview of the methods is provided below, while more detail can be found in the respective research papers.

Decomposition analysis

The decomposition of CO_2 emissions from fossil fuel combustion was carried out in the context of research on green growth policies (Paper 1). Its aim was to get a better picture of the potential impact of macroeconomic drivers on CO_2 emissions under a green growth policy regime. The decomposition was based on the Kaya-Identity (see also Box 1 in Section 3.2). By decomposing the total change of CO_2 emission from fossil fuel combustion in a given period into the activity effect (GDP), the energy intensity effect (primary energy supply per unit of GDP) and the carbon intensity effect of energy (CO_2 per unit of primary energy supply), an indication could be obtained of whether and how green growth policies have affected CO_2 emissions. The Logarithmic Mean Divisia Index (LMDI) was chosen as analytical tool for carrying out the additive decomposition (Ang, 2005).

Econometrics

Econometric methods were applied in three studies of this thesis. Firstly, a linear regression model was used to disentangle the drivers behind changes in CO_2 emissions from fossil fuel combustion (Paper 1). Secondly, econometric methods were used in the modelling of LCC of electric appliances in order to separate energy efficiency from other features influencing appliance prices (Paper 3). Thirdly, regression techniques were used to identify drivers behind people's WTP for emission reductions (Paper 5).

Scenario analysis

Scenario analysis is not a narrowly defined method but an umbrella term for a diverse set of approaches, ranging from literary narratives to quantitative modelling exercises (Swart, Raskin, & Robinson, 2004). What scenario analyses have in common is that they outline (and often compare) possible future developments under contrasting conditions. This is particularly important to address sustainability challenges such as climate change, which evolve in 'combined social and environmental systems under conditions of uncertainty, surprise, human choice and complexity' (Swart et al., 2004, p. 137) .

In this thesis, mainly quantitative scenarios were analysed, including scenarios of the Korean energy-economy system (Paper 1), and different policy scenarios to improve the energy efficiency of electric appliances (Paper 3). Moreover, people's acceptance and WTP in different carbon pricing scenarios were investigated (Papers 4 & 5).

Criteria-based indicator assessment

The assessment of indicators was used in this thesis research to investigate how indicator-based assessment performs in the case of public RD&D support for LCET, and to investigate what role the climate mitigation potential plays in the assessments of such policies (Paper 2). Assessment indicators were the evaluand in this case. The assessment of these indicators was in turn carried out based on a qualitative framework for indicator choice, which looked at the acceptance of an indicator, its ease of monitoring and its robustness (European Commission, 2005). While far from being objective, the systematic assessment of indicators for evaluation aimed to identify strengths and weaknesses of different approaches. Moreover, indicator assessment can show to what extent policy assessment reflects policy objectives including the need for rapid and deep decarbonisation.

Life-cycle cost modelling

The modelling of life-cycle cost (LCC) of energy consuming appliances plays a major role in the design of product policies (Paper 3). So far LCC modelling has been used to define MEPS that are economically beneficial for consumers (Buskirk, Kantner, Gerke, & Chu, 2014; Siderius, 2013). In this thesis an extended LCC model was used to investigate whether MEPS can capture the climate externality (as well as carbon pricing), and how carbon prices have to change in order to set the same LCC incentive as are reflected in a tightening of MEPS[9].

Contingent valuation

For some environmental goods, such as a stable climate or the reduction of CO_2 emissions, there is no straightforward market price. Contingent valuation (CV) is a method to investigate the non-market value of such environmental goods. It is a stated preference method (as opposed to revealed preference methods), which builds on surveys (Bateman, 2002). Typically, WTP for an environmental good or willingness to accept payment in exchange for the public good are researched. CV is driven by various different factors and is often very case and situation-specific. For this thesis CV was applied to research WTP for emissions reductions in different choice contexts (flying, car travel) and with different payment vehicles (climate surcharge on fuel and on air tickets, voluntary offsets), in order to gain behavioural insights with relevance for carbon pricing policies (Papers 4 & 5).

2.6. Methods for data collection

Due to the variety of methods applied and the interdisciplinary research approach, different data collection methods were required in the research of this thesis. This, in turn, enabled triangulation, that is, the cross-checking of results by using various data sources and methods (Mathison, 1988), which helped to make research results more robust. The main data collection methods for the studies included in this thesis are outlined below. Further details are provided in the appended research papers.

[9] Pages 3 ff. in Paper 3 provide further details about LCC modelling. The full methodology relies on previous work by one of the co-authors (Buskirk, Kantner, Gerke, & Chu, 2014).

Literature review

In order to sharpen the understanding of the research gaps (Section 1.2) and to create a conceptual framework for this thesis (presented in Chapter 3), initial literature reviews were carried out for both of the main research areas: green growth and behavioural insights. The literature review on green growth and the green (energy) economy included academic publications but also various reports from international organisations, such as the OECD and UNEP, which played an important role in shaping and promoting the green growth agenda. The literature review on behavioural insights focused on behavioural mechanisms relevant for economic decision-making in the context of LCET adoption and sustainable energy use. Due to the focus on economic decision-making, most of the conceptual background literature was found in the field of behavioural economics and its applications to the field of climate change and energy use. Literature from the fields of experimental economics and social and environmental psychology was consulted to a lesser extent.

Additional literature reviews were carried out to obtain specific contextual and methodological knowledge, including the topics policy evaluation, decomposition analysis, contingent valuation, technology innovation and directed technical change, RD&D financing of LCET, decarbonisation policies in cities, market failures in the energy system, carbon pricing, and climate policy for the aviation sector.

Interviews and consultations

Semi-structured interviews were conducted in order to get first-hand information from stakeholders (Barriball & While, 1994) when researching the evaluation practice of LCET support instruments (Paper 2). In addition to semi-structured interviews, various consultation meetings were carried out in order to discuss, test and refine the survey for the general public that aimed at investigating framing effects in decarbonisation policy design (Papers 4 & 5). These meetings included colleagues, students and people outside the university landscape, and their aim was to enhance the understanding of the survey by simplifying it, removing technical jargon and eliminating false presuppositions.

Database searches

Database research was carried out for two papers included in this thesis (Papers 1 & 3). In both cases data sources were made explicit and, as far as possible, data were made publicly available. For Paper 1, quantitative time series data from the International Energy Agency (IEA) were used as an input for the decomposition of CO_2 emissions changes into its main drivers. Additional data from statistics offices (in this case the Korean) were used to refine the decomposition and interpret its results.

In the case of life-cycle cost modelling of appliance markets (Paper 3), data from online market places in the UK and the US were used in the econometric and optimisation models. These data reflected the market offering at one point in time (2016). Access to the data of these market places was provided through one of the co-authors.

Survey

Surveys can help to obtain a statistical profile of a whole population (De Leeuw & Dillman, 2008). Therefore, they are a useful tool to find out characteristics and preferences of a small (but representative) sample with relevance for national climate policy. For the research of this thesis, an experimental online survey was conducted in order to elicit people's WTP for emissions reductions and test the effect of different payment vehicles (Papers 4 & 5). A challenge with surveys in general, and hypothetical choice settings in particular, is that they may contain various errors that potentially bias results (Groves & Lyberg, 2010). Hence, various efforts were made to reduce errors and mitigate biases, including test-runs of the survey, random sampling, the use of simple language, and screening out responses that were too quick.

2.7. Internal and external validity

Quantitative studies that are meant to inform policymaking have to take measures to ensure a high degree of internal and external validity (Sanderson, 2002). In order to strengthen the internal validity of research results, several approaches were taken in the research design for this thesis, including the triangulation of data sources (Paper 2), the use of different statistical approaches on the same data (Papers 1, 4 & 5), using a randomised within-subject design when investigating framing and context effects in the survey study (Papers 4 & 5), and conducting sensitivity analyses for key assumptions (Papers 3, 4 & 5). The sensitivity analysis for Paper 3 included key factors that may differ between energy systems, including the electricity price and the carbon intensity of the grid, and thereby contributed also to the external validity of results. The main approach to strengthen the external validity of findings was to focus on mechanisms rather than to give much weight to numerical results per se. Moreover, results were put into the context of findings from literature to see whether they challenge or support previous evidence.

2.8. Research Transparency

The reproducibility of scientific studies has been questioned in various fields, particularly in the social sciences including psychology (Johnson, Payne, Wang, Asher, & Mandal, 2016) and economics (Camerer et al., 2016), which are both research fields of this thesis. In order to facilitate the replication of empirical studies, the social sciences have started to copy research transparency practices from medical research.

For this thesis, several aspects concerning research transparency were implemented. Most importantly, quantitative data and files with the quantitative analysis were made publicly available wherever possible (Papers 3–5). Moreover, detailed method sections facilitate reproducibility. Besides data sharing and publishing data analysis sheets, another option to improve transparency and avoid 'p-hacking' and the non-reporting of null results is to pre-register studies and publish a pre-analysis plan. 'In pre-registration, researchers describe their hypotheses, methods, and analyses before a piece of research is conducted, in a way that can be externally verified.' (van 't Veer & Giner-Sorolla, 2016, p. 2). Various templates and platforms exist in order to formalise and verify pre-registration of analysis plans. For the research behind this thesis, pre-registration of a rough analysis plan was carried out for one empirical survey-study (Papers 4 & 5). The registration was carried out on the collaborative research platform 'Open Science Framework' (Sonnenschein & Smedby, 2017). This pre-analysis plan should, however, be considered a limited trial, as it was very short and did not contain much detail about the analytical approach. Besides the pre-analysis plan, the pre-registration, which was carried out before data collection started, also included the complete survey with all questions.

3. Conceptual framework

This chapter introduces the main concepts used in this thesis, including a generic climate change mitigation policy process (Section 3.1), the concept of green growth (Section 3.2) and behavioural insights (Section 3.3). The section on green growth deals with the focus on economic growth, de-coupling, technology optimism, market failures and economic instruments. The overview of behavioural insights includes an introduction to behavioural economics, an overview of behavioural insights in the context of LCET adoption and sustainable energy use, and policy approaches considering behavioural insights. Based on these concepts an overarching conceptual framework is introduced that ties together the research in this thesis (Section 3.4).

3.1. The process of climate change mitigation policy

In order to be able to assess green growth climate mitigation policies, it is useful first to distinguish the main steps of the general policy intervention process. Figure 3 outlines such a process in the context of climate change mitigation (based on Guedes Vaz, Martin, Wilkinson, & Newcombe, 2001; Neij & Åstrand, 2006). This process can also form the basis for criteria-based policy assessment, including criteria such as effectiveness, efficiency and relevance (see Section 2.3).

To start with, climate mitigation policy interventions have *objectives*, which may be driven by the Targets of the Paris Agreement and the respective NDCs. Policy objectives inform policy *inputs* (resources for the design and implementation of an intervention, including financing, staff and administration), which in turn drive policy *outputs* (the finalised intervention, such as new infrastructure, a subsidy scheme for LCET, energy standards, or a carbon pricing instrument) (Guedes Vaz et al., 2001). Policy outputs are meant to have tangible *outcomes* in society or economy, e.g. the adoption of LCET (Neij & Åstrand, 2006). Finally, policy outcomes drive social and environmental *impacts*. To be relevant, a policy intervention should address overarching societal needs and targets, which – in the case of climate change – are well captured in the Paris Targets. Whether an intervention was or can be expected to be effective is assessed by comparing outcomes and/or impacts to the initial objectives. It is, however, the final impact towards solving the challenge of climate change that shows whether an intervention was adequate.

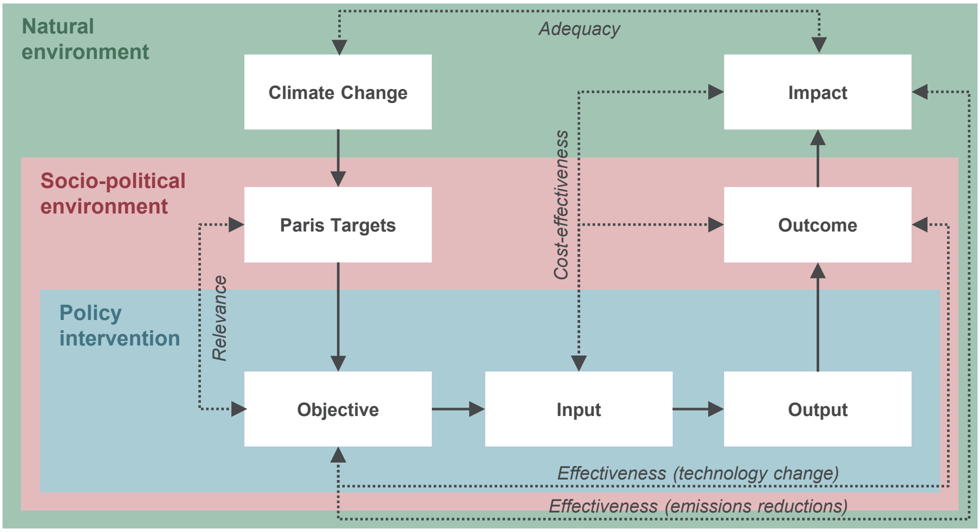

Figure 3
Schematic overview of the policy intervention process and associated evaluation criteria (adaptation of Fig.6 in Guedes Vaz et al., 2001).

What becomes clear when looking at Figure 3 is that evaluation criteria as such cannot explain the 'arrows between the boxes'. Therefore, a key task in assessing an intervention is to identify and explain the *mechanisms* that lead from initial policy objectives to outcomes and impacts (Astbury & Leeuw, 2010). Studying the main causal mechanisms that are expected to drive policy outcomes has been suggested as a viable approach for policy assessment and an alternative to policy trials that test-run whole interventions at smaller scale (Ludwig, Kling, & Mullainathan, 2011). While Figure 3 outlines the generic climate policy process that underlies the research for this thesis, the whole policy process was not assessed in any of the individual papers. The specific objectives, inputs, outputs, outcomes and not least mechanisms of the respective papers are further elaborated in Section 4.

3.2. Green growth

Throughout the past decade, and particularly after the 2008/09 global financial crisis, green growth policies have drawn the interest of international institutions such as OECD, UNEP, the World Bank, the European Commission, and the Nordic countries (Stoknes & Rockström, 2018). Several countries introduced green fiscal stimulus policies after the crisis, including China and Korea. Recently, the idea of yet another 'Green New Deal' as a climate change mitigation strategy was picked up by Democratic Party members in the US, suggesting it be moved to the centre of their 2020 electoral campaign strategy (Ocasio-Cortez, 2018). At the international level, 'Green Economy' initiatives were championed by UNEP (UNEP, 2014), the OECD (OECD, 2011b) and the World Bank (World Bank, 2012), which together established the Green Growth Knowledge Platform (GGKP). Albeit not new, the concept of green growth (Barbier, 2010a; Barbier & Markandya, 2013; OECD, 2011b; Pearce, Markandya, & Barbier, 1989; UNEP, 2009) is defined among others by the OECD:

> Green growth means fostering economic growth and development while ensuring that natural assets continue to provide the resources and environmental services on which our well-being relies.

There is no agreement on this or other green growth and green economy definitions (Bowen & Hepburn, 2014; Jacobs, 2012; Mundaca et al., 2016). Some are very inclusive and resemble definitions of sustainable development (UNEP, 2011), while others are more focused on the environmental domain (OECD, 2011b). When reviewing the literature, it becomes clear that, despite the lack of a common definition, there are some reoccurring core aspects of the green growth concept and associated policy approaches. These aspects include the focus on economic growth, the belief in de-coupling economic growth from environmental impacts, optimism in technology progress, and the policy focus on removing market failures. All of these aspects are further explained below.

Growth focus

The first key defining element of green growth is the focus on economic development, aka GDP growth. In the standard version of green growth it has been argued that acting on climate change is compatible with economic growth as the costs of inaction will be higher than the costs of mitigation (Jacobs, 2012; N. H. Stern, 2006). On the other hand, it was criticised that mitigation costs occur today (and potentially slow down growth), whereas benefits are spread far into the future. In the context of the 2008/09 global financial crisis, a time when policies with the reputation to slow down growth were not popular, a stronger version of green growth emerged. The strong version of green growth claimed that environmental policy may be a driver of economic growth (Jacobs, 2012). This argument is backed by a meta-assessment of policy assessments that concluded 'whatever the underlying approach of green economy is, it stresses the importance of integrating economic and environmental policies in a way that highlights the opportunities for new sources of economic growth' (Speck & Horton, 2011, p. 93).

At the conceptual level, the growth focus of green growth approaches is fundamentally questioned by research concerning de-growth (Kallis, Kerschner, & Martinez-Alier, 2012), sustainable consumption (Lorek & Spangenberg, 2014) and sufficiency (Princen, 2003). A recent review of the growth versus climate debate, suggests a third way besides green growth and de-growth, namely 'agrowth' (van den Bergh, 2017). Agrowth means shifting the focus away from growth all together. In a similar manner, it has been argued that neither green growth nor de-growth are suitable concepts to inform policy as they show flaws in their conceptualisations of social welfare (Jakob & Edenhofer, 2014).

De-coupling

> Decoupling economic activity from its ecological consequences is central to the goals of international sustainability and development. It's the foundation of American faith that technology can resolve climate change without the need to substantially change our life styles. It's the holy grail of 'green growth.' (MacKinnon, 2017)

The above-mentioned growth-orientation of green growth approaches is only compatible with a reduction in environmental impacts if these impacts can be de-coupled from economic activity. The key metrics for this de-coupling in the context of climate change is the carbon intensity of the economy (Bowen & Hepburn, 2014), i.e. the average amount of GHG emissions per unit of economic output, often measured in carbon dioxide equivalents (CO_{2-eq}) per EUR or USD of economic output. The carbon intensity of the economy can be further broken down into sub-drivers, including the energy intensity of the economy and the carbon intensity of energy supply (see Box 1 below).

It is important to note that improvements in these factors do not necessarily imply absolute reductions in GHG emissions, but may simply mean that the emissions increase at a slower pace than GDP, which is referred to as *relative de-coupling* (see Figure 5a). In contrast, the concept of *absolute de-coupling* goes beyond the mere improvements of carbon intensity (of the economy) and means concurrent economic growth and absolute reductions of GHG emissions (see Figure 5b).

Box 1
The Kaya Identity

In the field of climate change the **Kaya-Identity** (Kaya, 1990) is a specific expression of the IPAT-equation (Ehrlich & Holdren, 1971), which decomposes CO_2 emissions into energy-economy-environment drivers (see Figure 4).

GDP as an indicator of economic activity measures the monetary value of final goods and services that are bought by the final user on a market (e.g. a country). As such, GDP provides information about the volume of economic activity, but not about its distribution, its origin, or its environmental impacts. Economic activity is associated with resource and energy use, which in turn cause GHG emissions. Other things being equal, an increase in economic activity results in an increase in GHG emissions.

GDP growth can be partly offset by improvements in the **energy intensity of the economy**, which is the ratio of energy inputs, often measured in total primary energy supply (TPES), to GDP outputs. The effect of changes in energy intensity can be broken down further into two sub-drivers. First, **structural change of the economy**, e.g. the shift from industrial activity to services (tertiarisation), is often regarded a background trend that can – if at all – only be influenced by long-term comprehensive policy interventions. Moreover, structural change might only result in carbon leakage, i.e. the shift of emissions from heavy industry to other countries. The **energy intensity within specific sectors**, on the other hand, depends largely on the energy efficiency of the production processes and technologies. Energy intensity differs significantly between sectors (see for Sweden Karimu, Brännlund, Lundgren, & Söderholm, 2017). Besides industry and manufacturing, the household sector can also contribute to energy intensity improvements.

The **carbon intensity of the energy mix** is the ratio of CO_2 emissions to energy use (e.g. TPES). This ratio is also influenced by two sub-drivers. The **share of fossil fuels** decreases as renewable energy enters the mix. This can be in the form of renewable energy sources for electricity generation (RES-E), such as wind, solar or hydro, or in the form of bioenergy. High hopes are also put into combining bioenergy with carbon capture and storage (BECCS), one of the CDR options, in order to reduce the carbon intensity of energy. The **carbon intensity of the fossil share** of the energy mix can be reduced by implementing CCS technology or by fuel-switching (e.g. from coal to gas).

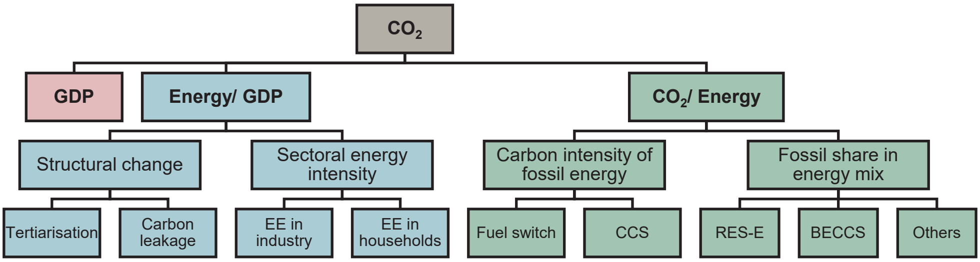

Figure 4
Overview of CO_2 emission driver (adapted from Peters et al., 2017). The second row displays the Kaya Identity, which is extended in the third row. The fourth row gives an overview of specific trends and technologies (EE = energy efficiency, CCS = carbon capture and storage, RES-E = electricity from renewable energy sources, BECCS = bioenergy CCS).

It has been highlighted that even under the notion of absolute de-coupling, emissions are not necessarily de-coupled from GDP (Cohen, Jalles, Loungani, & Marto, 2018). An increase in GDP may still result in a slower decrease of CO_2 emissions, as shown in Figure 5b, where GDP and CO_2 are perfectly correlated, in other words 'coupled'. In this context *actual de-coupling* would mean that the association between GDP and CO_2 is weakened (see Figure 5c). This is different from absolute de-coupling as in absolute de-coupling the association between GDP and CO_2 might still be strong, even though they develop in different directions.

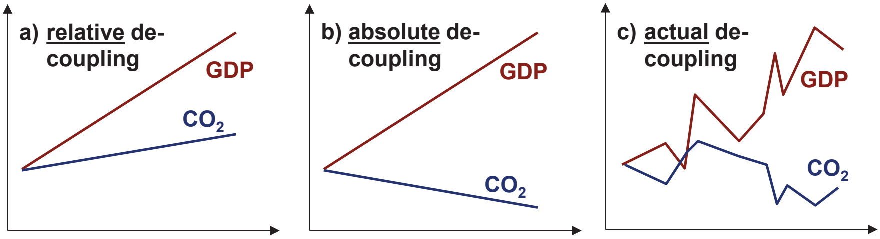

Figure 5
Schematic Overview of different conceptualisations of de-coupling.

Criticism of green growth centres around the potential for (absolute) de-coupling (Antal & van den Bergh, 2014). While several countries achieved absolute reductions of domestic emissions after 2005 (Le Quéré et al., 2019), these reductions have in several cases coincided with economic crises. Globally, emissions briefly decreased during the 2008/09 financial crisis but quickly rebounded afterwards (Peters et al., 2012). Occurrences of absolute de-coupling are even less frequent if consumption-based emissions are considered. A study of 29 high-income countries between 1991 and 2008 did not find evidence of de-coupling for consumption-based emissions (Knight & Schor, 2014). More recently there are some instances of de-coupling consumption-based emissions from economic growth (Le Quéré et al., 2019 supplementary information) but both territorial and consumption-based emissions reductions 'fall a long way short of the deep and rapid global decarbonization of the energy system implied by the Paris Agreement temperature goals' (Le Quéré et al., 2019, p. 217).

Technology optimism

The de-coupling that is envisioned by green growth climate policies relies heavily on technology change and the large-scale adoption of LCET (Popp, 2012). Considering both the scale of economic growth needed for developing countries to 'catch up' and the emissions reductions implied in the Paris Targets, an optimistic view of technology development is needed to believe that technology change will be sufficient to drive the required de-coupling. This technology optimism inherent in the green growth paradigm is closely related to technological optimism in the ecological modernisation governance paradigm (Bäckstrand & Lövbrand, 2007) and technological optimism in the climate mitigation modelling and policy discourse (Arvesen, Bright, & Hertwich, 2011; P. Smith et al., 2016), which does not fully account for uncertainties in technology forecasting and associated cost increases (Yeh & Rubin, 2012).

Market failures

In order to drive technology change, it is clear that new LCETs have to be developed, LCET innovations have to be commercialised, and mature LCETs have to be widely deployed (Blanco et al., 2014; Rogelj, Shindell, et al., 2018). It has been argued that with respect to each of these steps market failures hinder progress. Relevant market failures include unpriced externalities (CO_2 emissions), R&D knowledge spillovers of LCET innovation, the principal-agent problem (e.g. for energy efficiency investments in tenant-occupied buildings), and various lacks of information (Gillingham et al., 2009). Knowledge spillovers imply that companies invest less in LCET innovation than socially optimal as they cannot reap the whole benefits triggered by their investments. R&D knowledge spillovers are one of the main reasons why some environmental economist argue for a policy mix that goes beyond carbon pricing (C. Fischer & Newell, 2008). Beside the (largely) unpriced climate externality, the greatest market failure the world has ever seen (N. H. Stern, 2006), this thesis mainly deals with R&D knowledge spillovers of LCET innovation (Paper 2) and the lack of information about LCC of energy efficient appliances (Paper 3).

Economic instruments

As under the green growth concept, great emphasis is put on the functioning of the market (for environmental technologies), policies addressing market failures, particularly economic instruments, are another central aspect of the green growth concept (Jacobs, 2012; Jouvet & de Perthuis, 2013; Reilly, 2012). Even though a specific definition or taxonomy of green growth policies is lacking, a focus on economic instruments rather than regulation can be found in several publications on the topic[10]. The latest IPCC assessment report describes economic instruments as follows (Kolstad et al., 2014, p. 239):

> Economic (or market) instruments include incentives that alter the conditions or behaviour of target participants and lead to a reduction in aggregate emissions. In economic policy instruments, a distinction is made between 'price' and 'quantity'. A tradeable allowance or permit system represents a quantity policy whereby the total quantity of pollution (a cap) is defined, and trading in emission rights under that cap is allowed. A price instrument requires polluters to pay a fixed price per unit of emissions (tax or charge), regardless of the quantity of emissions.

[10] Regulations and administrative instruments are sometimes even mentioned as a green economy barrier. The latest OECD chapter on green growth included an indicator to monitor environmental policy as a potential *burden* for market entry and competition (OECD, 2018).

Among the economic instruments considered in the context of green growth, carbon pricing instruments (both quantity and price based) stick out as the ones highlighted most (Jones, 2011; Jouvet & de Perthuis, 2013; OECD, 2011a, 2013).

From a theoretical perspective, carbon pricing instruments are frequently justified by the external costs associated with carbon emissions. This externality is frequently called the social cost of carbon (SCC; see Box 2). The economically efficient tax level should be set so that the SCC equals the marginal abatement costs (MAC). In practice, however, considerable uncertainties about MAC and SCC make efficient taxation challenging. Moreover, political and public acceptability and lobbying heavily influence the carbon price levels that are actually implemented.

Box 2
Overview of concepts for monetizing a tonne of CO_2 emissions (see also Duong, 2009; Rogelj, Shindell, et al., 2018).

Social Cost of Carbon (SCC) 'measures the total net damages of an extra metric tonne of CO_2 emissions due to the associated climate change […]. Negative and positive impacts are monetized, discounted and the net value is expressed as an equivalent loss of consumption today' (Rogelj, Shindell, et al., 2018, p. 150). The estimation of SCC is challenging and based on value judgements, for instance 'how non-market damages and the distribution of damages across countries and individuals and between current and future generations are valued' (Rogelj, Shindell, et al., 2018, p. 150).

Marginal Abatement Cost (MAC) measures the cost of reducing one more unit of pollution. In the case of climate change, this is typically one tonne of CO_2 emissions. The MACs of different technologies are often summarised in MAC curves. The MAC concept and MAC curves depend on many assumptions and have several limitations (Kesicki & Ekins, 2012).

Shadow Price of Carbon is the marginal abatement cost of carbon that is associated with a certain climate goal under consideration, such as the goals of limiting warming to 1.5°C or well below 2°C (Rogelj, Shindell, et al., 2018, p. 150).

(Explicit) Carbon Price 'The price for avoided or released carbon dioxide (CO_2) or CO_2-equivalent emissions. This may refer to the rate of a carbon tax, or the price of emission permits. In many models that are used to assess the economic costs of mitigation, carbon prices are used as a proxy to represent the level of effort in mitigation policies.' (Matthews, 2018, p. 9). An empirical overview of explicit carbon pricing instruments, their price levels, and their scope is provided in the annual World Bank publication 'State and Trends of Carbon Pricing' (World Bank & Ecofys, 2018).

Willingness to Pay (WTP) per tonne of CO_2 expresses a (hypothetical) price at or below which people are willing to offset their personal carbon emissions. In other applications, but closely related, it may also express people's WTP for climate change mitigation. WTP is often measured per tonne of CO_2, but there are also studies investigating total WTP per year or WTP per unit of carbon-emitting activity (e.g. flight) exist.

As can be seen from the above, policies under the green growth concept put the focus on LCET markets and their failures and highlight the importance of carbon pricing. Green growth is largely informed by concepts and theories from neoclassical economics (Reilly, 2012). Behavioural insights do not appear to play a major role in this. The next section presents behavioural insights that can be considered in policy assessment, design and implementation, and have an impact on LCET adoption.

3.3. Behavioural insights

Should the facts be allowed to spoil a good story? (Lovell, 1986)

The green growth concept tells a compelling story, namely that with the right choice of economic policy instruments rational market actors will overcome market failures, increase their adoption of LCET and, thereby, both economic growth and emission reductions will be achieved. The story, however, is not without its flaws and this section provides the conceptual background for challenging and further developing the decision-making model underlying the functioning of green growth policies.

This section starts by providing some background to economically rational decision-making. The rational choice model is challenged heavily by empirical research findings from behavioural sciences. Besides psychology, one field that has particularly challenged rational choice is behavioural economics, which is the main research stream used for this thesis. It is briefly introduced below. Following this intro, behavioural insights are further defined and an overview of insights with relevance to LCET adoption is provided. Finally, policy approaches considering behavioural insights are introduced.

Rational choice theory

The key assumptions behind the choices of a rational actor are described well by one of the major critics of the simplified economic view on decision-making, the economist Herbert Simon (Simon, 1955):

> This man is assumed to have knowledge of the relevant aspects of his environment which, if not absolutely complete, is at least impressively clear and voluminous. He is assumed also to have a well-organized and stable system of preferences, and a skill in computation that enables him to calculate, for the alternative courses of action that are available to him, which of these will permit him to reach the highest attainable point on his preference scale.

To paraphrase, if people have access to almost all information, if they have consistent and stable preferences, and if they are utility maximisers, then the market mechanism will lead to an efficient outcome, that is an outcome in which resources are used in the best possible way and nothing is wasted. In this model, individual decision-making can be affected by simply adjusting the perceived costs and benefits of a choice (Camerer, 1999).

Climate policy based on rational choice puts the focus on removing market failures so that rational actors choose the low-carbon options more frequently (Shogren &

Taylor, 2008). That involves, first and foremost, pricing the climate externality, so that the social costs associated with carbon emissions are reflected in market transactions (Gowdy, 2008). Besides the climate externality there are further market failures that have been identified in the climate context (see Section 3.1). It is important to note, however, that removing market failures is by no means an easy task. Market prices that reflect the climate externality would require comprehensive carbon pricing interventions at a high price level (de Coninck et al., 2018; Rogelj, Shindell, et al., 2018).

While implementing effective climate change mitigation policy, assuming rational choice decision-making is a considerable challenge, it is likely based on a flawed model of decision-making. The rational choice model has been criticised from various angles, among others from a behavioural economics perspective (Camerer, 1999; Kahneman, 2003).

Behavioural economics at a glance

Behavioural economics is a research stream that is critical of rational choice. In contrast to rational choice theory, behavioural economics starts off from the assumption that 'economic behavior can be understood by observing the actions of actual individual agents and the structural context within which they operate' (Gowdy, 2008, p. 637). From a methodological point of view, behavioural economics is based on empirical and inductive research (Camerer et al., 2011; Thaler, 2015), which is reflected in the methods that are used, including lab, field and natural experiments, surveys, and computer simulations (Tomer, 2017).

Through various empirical studies it could be shown that systematic deviations from (economically) rational decision-making occur in several choice situations. These patterns are often referred to as *behavioural anomalies* (Gillingham and Palmer, 2014) that explain why people fail to behave as predicted by the rational choice model (Shogren and Taylor, 2008). If behavioural anomalies imply a difference between decision utility at the time of choice and the utility eventually experienced after the choice (Kahneman & Thaler, 2006) they are called *behavioural failures* (Gillingham and Palmer, 2014)[11]. The more systematic these behavioural anomalies and failures are, the stronger are the behavioural insights that can be gained for policy-making.

Behavioural economics is a diverse field rather than a unified theory. Already in its criticism of rational choice theory different aspects are highlighted, including the bounded rationality, bounded willpower, and bounded selfishness that can be observed in economic decision-making situations (Mullainathan & Thaler, 2000).

[11] The difference between decision utility and experienced utility is sometimes also referred to as 'internality', that is the (future) individual costs and benefits that are not factored into a decision (Allcott, Mullainathan, & Taubinsky, 2014; Herrnstein, Loewenstein, Prelec, & Vaughan, 1993).

Related to these limitations of rational choice, several different branches of behavioural economics can be identified including among others prospect theory, intertemporal choice, norm-based behaviour, and heuristics (Brekke & Johansson-Stenman, 2008; Pollitt & Shaorshadze, 2013).

Prospect theory deals with the subjective value of losses and gains compared to a reference point (Kahneman & Tversky, 1979; Tversky & Kahneman, 1992). Several behavioural anomalies have been explained with prospect theory, including reference-dependent preferences, the endowment effect, loss aversion and the status quo bias (Kahneman, Knetsch, & Thaler, 1991). Closely related is the area of heuristics. Heuristics are simplified and/or intuitive decision-making rules that may be quick and practical but can lead to suboptimal choices (Slovic, Finucane, Peters, & MacGregor, 2002; Tversky & Kahneman, 1974). Satisficing, for example, means that people, when faced with multiple options, decide in a way that gives them a satisfactory result rather than trying to maximise their utility (Kahneman, 2003; Simon, 2008).

Besides heuristics and biases, another area of behavioural economics deals with preferences in intertemporal choice. People often have a preference for the time when costs and benefits are realised and value future outcomes differently than present (Frederick, Loewenstein, & O'Donoghue, 2002; Loewenstein & Thaler, 1989). Due to myopia and self-control problems, people tend to discount future gains from current investments (e.g. in energy efficient appliances) at rates that are much higher than market discount rates (J. A. Hausman, 1979; Thaler, 1981).

Finally, preferences are also influenced by personal values and social norms. The latter are formal or informal social patterns that influence the behaviour of a given group (Abrahamse & Steg, 2013). At least two categories of social norms can be distinguished. First, descriptive norms are the perception of what most people do in a given situation; second, injunctive norms describe the degree to which a certain behaviour is generally approved or rejected (Reno, Cialdini, & Kallgren, 1993). Personal values that affect preferences can either conform with or diverge from social norms. In the environmental context a differentiation has been made between egoistic, altruistic, hedonic and biospheric values (Steg, 2016), which can all be held by individuals to varying degrees.

Behavioural insights for climate change mitigation

> To successfully […] change behaviours that contribute to climate change […] it is necessary to consider individual capabilities, cognitive processes, biases, values, beliefs, norms, identities and social relationships, and to integrate understanding at this level into broader understanding of human interactions with a changing climate. (Clayton et al., 2015, p. 644)

The previous section illustrated that behavioural economics (and other behavioural sciences) do not built on a unified theory but draw from a variety of disciplines, concepts and methods. Accordingly, the insights gained from these fields for policy-making are also difficult to organise in an overarching framework (OECD, 2017a). As summarised in the quote above, various internal and external factors need to be considered to effectively understand, analyse and change behaviours (P. Stern, Sovacool, et al., 2016).

While classification and conceptualisation are challenging, behavioural insights can be defined as the operationalisation of findings from behavioural-oriented disciplines (e.g. behavioural economics) and other behavioural sciences (e.g. environmental psychology, sociology) in a policy context (OECD, 2017c). It is important to note that behavioural insights do not set the objectives for policy-making (as green growth does), but instead inform and help in choosing and designing policies once societal or environmental goals are decided.

In order to consider behavioural insights in policy assessments, it is useful to differentiate between moderators and mechanisms (see Figure 6). *Moderators* are (behavioural) factors that are not affected by a policy intervention but that affect the extent of policy outcomes; *mechanisms*, in contrast, establish a causal connection between a policy intervention and its outcomes (Ferraro & Hanauer, 2014). Moderators include various socio-demographic and psychological variables (Abrahamse & Steg, 2011), while mechanisms also include, besides rational choice, other reoccurring behavioural patterns, including heuristics, time-dependent preferences and norm-based behaviour (Pollitt & Shaorshadze, 2013).

The emissions reduction potential associated with behavioural insights is a function of both technical factors (how much less polluting are alternative technologies or practices) and behavioural plasticity (the share of people that can be induced to act) (Dietz, Gardner, Gilligan, Stern, & Vandenbergh, 2009). In other words, people have to adopt and use LCETs; and to make them do that, their behavioural particularities need to be considered. In the field of LCET, the short-term mitigation potential through changes in household behaviour has been estimated to be about 20% in the US (Dietz et al., 2009) and up to 16% in the EU (van de Ven, González-Eguino, & Arto, 2018).

There are several examples for climate change mitigation interventions based on behavioural insights (Frederiks, Stenner, & Hobman, 2015), including green defaults (Dinner, Johnson, Goldstein, & Liu, 2011; Pichert & Katsikopoulos, 2008; Sunstein & Reisch, 2014), real-time feedback to households on energy use (Darby, 2006; C. Fischer, 2008), and approaches to reduce electricity use based on social and community norms (Allcott, 2011). It is important to note that the functioning of (climate) policy interventions based on behavioural insights is context-specific and depends on various mediating factors (Abrahamse & Steg, 2011; P. Stern, Sovacool, et al., 2016). Contextual factors include for instance the individual choice setting, time, place and social institutions.

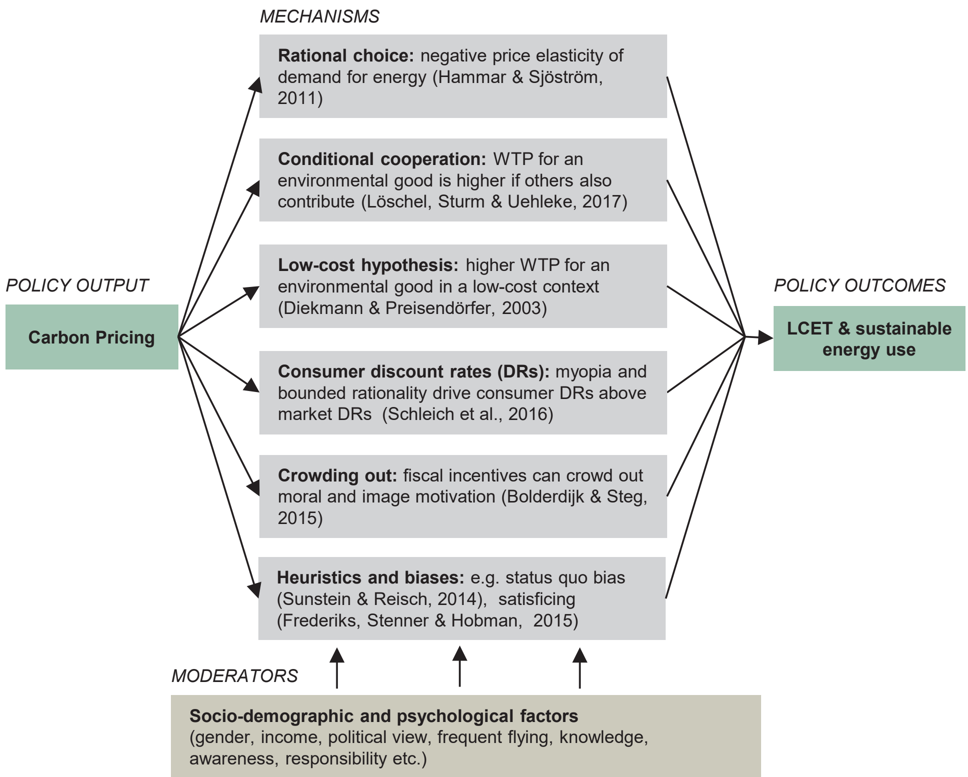

Figure 6
Overview of behavioural mechanisms and moderating factors influencing the outcome of carbon pricing (adapted from Ferraro & Hanauer, 2014).

For the research presented in this thesis, Figure 6 exemplifies some of the behavioural mechanisms that are relevant in the context of carbon pricing. The listed mechanisms all have support from literature and will be further discussed against the background of findings from the research papers (Chapter 5). Note that the mechanisms are not mutually exclusive and may overlap. Insights about such behavioural mechanisms and moderators need to be taken into consideration when designing green growth climate policies.

Using behavioural insights to design green nudges and other (climate) policies has been critically debated from an ethical perspective (Schubert, 2017), with libertarians being particularly critical towards the paternalistic State that makes decisions on behalf of people (D. M. Hausman & Welch, 2010; Mitchell, 2004). However, in the context of green growth climate policies, which mainly concern the energy system, using behavioural insights should not be considered problematic from a libertarian perspective because the systems for both energy generation and consumption are 'massively architectured' to start with (Kasperbauer, 2017). Most energy-related choices are shaped by State interventions anyway, whether the State uses behavioural insights or not. Moreover, libertarianism is not the only ethical perspective towards using behavioural insights in policy-making. For other ethical positions (e.g. utilitarianism) it is more important what nudges are used for and whether nudges enhance welfare (Sunstein, 2015), which is likely the case for many climate change mitigation policies.

3.4. A combined framework for policy assessment

The overarching conceptual framework of this thesis was developed by populating the general policy process for climate change mitigation (as presented in Section 3.1) with core aspects of the green growth concept (Section 3.2) and integrating the insight that behavioural mechanisms affect the functioning of policy interventions (Section 3.3). Pictured in Figure 7, the framework accounts for the two main research objectives of this thesis and incorporates the concepts of green growth and behavioural insights. The right column shows the focal aspects of policy assessment, including policy objectives, outputs, their outcomes and eventually policy impacts (compare Figure 3 above).

On the left side, the policy process of green growth policy interventions is stylised. Green growth's dual objective of economic growth and decarbonisation informs the choice of green growth policy outputs, which are typically focused on economic instruments, including carbon pricing and subsidies. These interventions, by addressing market failures, target the development and adoption of LCET and sustainable energy use. Increased adoption of LCET and sustainable energy use are supposed to lead to climate change mitigation impacts. For this to happen, the crucial mechanism is (absolute) de-coupling.

In order to account for the second research objective, the framework also considers behavioural insights (see Section 3.3). Unlike green growth, behavioural insights are not associated with specific policy objectives and, hence, do not favour a certain outcome or impact. By taking into account behavioural insights, behavioural mechanisms can be addressed in the design of policy outputs (here green growth climate mitigation policies) so that they more effectively lead to climate change mitigation.

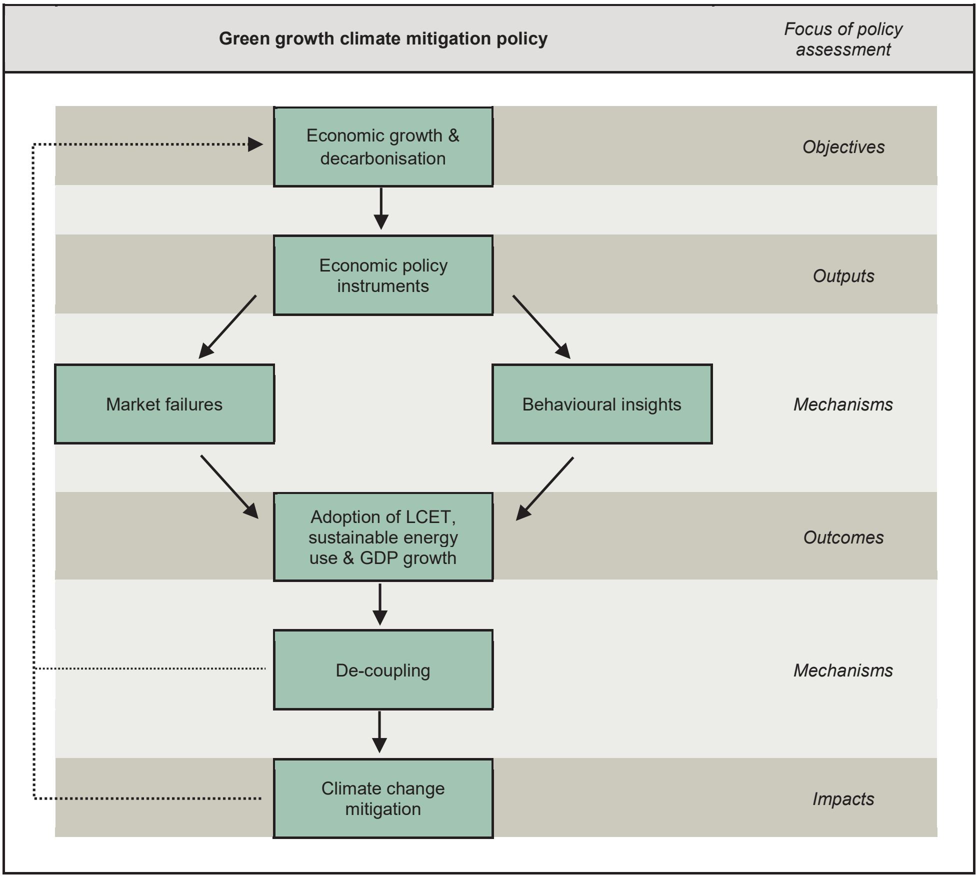

Figure 7
Conceptual framework for the assessment of green growth climate mitigation policies.

4. Results

This chapter aims to give an overview of the research findings, which are then (in Chapter 5) related to the existing literature and discussed with respect to the research questions. The chapter is organised in accordance with the research papers that form this thesis and includes the background of the respective study, its connection to the overarching conceptual framework, key results and their relation to the research questions, and finally policy implications. More detailed results and the sensitivity analyses of several results can be found in the appended papers.

4.1. Response of key CO_2 emissions drivers to a national green growth programme

Background

Paper 1 deals with Korea's Green Growth Strategy (2009-2013), which was enacted in response to the 2008/2009 global financial crisis and included public spending with an overall volume of nearly USD 100bn. Besides economic recovery, the strategy targeted CO_2 emissions reductions of 30% against BAU by 2020. The analysis of this policy in Paper 1 combined a decomposition analysis of CO_2 emissions from the energy system and econometrics with a review of energy and climate policies, including related structural changes.

Conceptual framework

The policy analysis provided in this paper followed the green growth climate policy process outlined in Figure 7. The analytical focus was on policy *outcomes* (development of GDP, energy intensity of the economy, and carbon intensity of energy) and whether the *mechanism* of de-coupling leads to the desired *impact* (climate change mitigation). To the extent possible the results were connected to the specific policy *output*, that is, public spending under the Five-Year Plan for Green Growth for different LCET and infrastructure.

Key results

The results of both econometric analysis and additive decomposition suggest that Korea was not successful in achieving the aspired impact of stable CO_2 emissions, let alone CO_2 emissions reductions. In contrast, Figure 8 shows that compared to the pre-crisis year 2008, changes in energy intensity and carbon intensity even contributed to higher (!) emissions in the first four years under the Green Growth Strategy. These drivers were added to the strong effect that increased economic activity had on emissions growth. Hence, Korea's Green Growth Strategy was not successful in achieving absolute de-coupling.

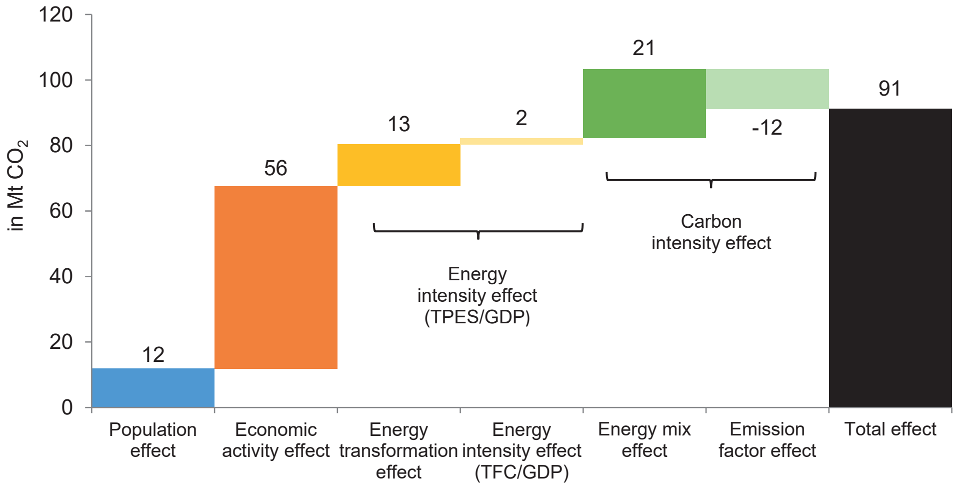

Figure 8
Additive LMDI decomposition analysis of changes in CO_2 emissions from fossil fuel combustion in Korea 2008-2012. Economic activity was based on GDP per capita; energy transformation based on TPES/ TFC; energy intensity based on TFC/ GDP; energy mix on shares of different fossil fuels; emission factor based on specific CO_2/ TPES for the respective fuels.

This clear outcome can be partly explained by the specific policy outputs under the Green Growth Strategy. To start with, only 26% of public spending was actually dedicated to LCET. Moreover, accompanying policies between 2008 and 2012 were either insufficient (support policies to renewable energy), counterproductive (subsidies to fossil fuel exploration and production, publicly subsidised electricity tariffs and decreased share of excise duty in the fuel price) or not yet implemented (emissions trading system launched in 2015).

One challenge of studying the outcomes of Korea's Green Growth Strategy was the potential time lag between investments in LCET and their effect on carbon emissions. For Paper 1 data was only available until 2012, the fourth year of Korea's 2009-2013 Five Year Plan for Green Growth. In order to investigate more recent effects for this thesis, the latest data available from the IEA (2013–2016) was used to update the decomposition analysis and investigate more recent developments (see Figure 9).

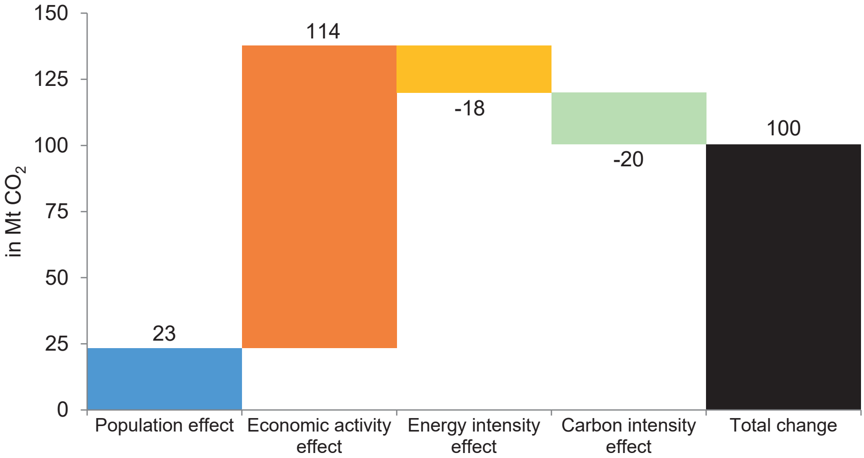

Figure 9
Results of additive LMDI decomposition of CO_2 emissions from fuel combustion in Korea for the period 2008–2016.

When comparing Figure 8 to Figure 9 it becomes clear that in the second half of the 2008–2016 period both population and economic growth remained strong drivers of emissions growth in Korea. In contrast, both energy intensity and carbon intensity improved so that they mitigated even stronger emissions growth. This suggests that the Green Growth Strategy and accompanying policies might have been at least partially successful in counterbalancing some of the additional emissions associated with economic growth.

With respect to RQ 1, these results question the extent to which green growth has been a suitable policy strategy for short to mid-term decarbonisation of the economy, particularly considering the strong association between economic activity and CO_2 emissions that could not be broken up by improvements in technology. Moreover, and concerning RQ 2, the results of Paper 1 show that objectives of green growth policies are not limited to climate change mitigation and may even include seemingly detrimental objectives such as the promotion of 'new growth engines' or energy security (via exploration of fossil fuel deposits).

Policy implications

- Reorient economic policies from GDP growth to improvements in well-being, job creation, and a structural change to the economy (e.g. by lowering labour costs relative to energy and carbon prices).
- Ensure a significant and stable price on carbon to improve the efficiency and carbon intensity of power generation; and increase electricity tariffs by introducing market pricing and taxes in order to manage demand and incentivise energy efficiency.
- Ensure a stable and reliable support scheme for renewable energy and re-evaluate the role that nuclear power can play in the energy mix.
- Give incentives for the purchase of low-emission vehicles and progressively tax vehicles with high CO_2 emissions.

4.2. Indicator choice for the assessment of RD&D financing of LCET

Background

Reaching the Paris Targets will require additional RD&D of LCET. States support RD&D with various financing instruments. The assessment of these instruments and the policy design implications that follow from such an assessment depend among others on the choice of indicators for evaluation. As success of public RD&D is not guided by concerns for the climate alone, different stakeholders may have their own criteria or indicators for success in this context. Paper 2 investigated indicators for the evaluation of LCET-specific RD&D support policies in the Nordic countries, which are frequently mentioned as leading countries with respect to eco-innovation. As a first step, indicators were listed. Then the list was analysed against several criteria[12]. Moreover, the impact of indicator choice on the design of public RD&D financing for LCET was analysed.

Conceptual framework

The policy analysis provided in this paper followed the green growth climate policy process outlined in Figure 7. At the centre of this paper is the analysis of policy

[12] See also Miremadi, Saboohi, & Jacobsson (2018) who refer to Paper 2 and expand the list of indicators, before also applying a set of value criteria to this list.

objectives of public RD&D support for LCET and how objectives may influence *outputs*, that is the design and focus of the specific RD&D financing instruments. However, the paper does not focus on the explicit policy objectives of these interventions but on the objectives that are actually used in their assessment and that are perceived as central by the people who administer the respective support instruments.

Key results

From a methodological perspective, the study of RD&D financing of LCET in the Nordic countries showed that 'a structured assessment of indicators can help to point up the trade-offs and limitations that are inherent in indicator-based evaluation. Selecting indicators can introduce bias'. As a matter of fact, no indicator or set of indicators emerged as clearly superior from the analysis, which implies there is room for influencing evaluation results by choosing certain indicators over others.

Besides this finding about indicator assessment, the analysis revealed three major trends reflecting policy objectives. Firstly, a focus on short-term economic performance and return on investment (ROI) was discovered. For most of the analysed instruments, their impact on turnover and exports was highlighted, while especially for public VC financing instruments even ROI targets were mentioned. Secondly, a strong emphasis on the additionality of financing for RD&D in LCET was found. The importance of additionality appeared to be largely driven by concerns about compliance with European State Aid regulation rather than genuine concerns that public financing might crowd out private investments. Thirdly, it became clear that support schemes for RD&D in LCET largely disregard decarbonisation potential. While mentioned in the official documentation of support instruments (e.g. legal documents and websites), the climate change mitigation potential did not play a role in actual assessments.

These results from Paper 2 contribute to answering RQ 2 by showing that the objectives of green growth-related policy interventions are not always focused on climate change mitigation. In contrast, they might even largely dismiss decarbonisation potential and focus on support for those LCETs with large economic potential. These findings also contribute to RQ 1, because innovation policies that largely ignore decarbonisation potential might not contribute as much to climate change mitigation as possible. Moreover, the time lag between RD&D support to LCET and the large-scale adoption of these (new) technologies implies that green growth policies targeting technology development (while crucial in the long run) are not a suitable driver for short to mid-term decarbonisation.

Policy implications

- A bias towards ROI and the (short-term) development of jobs, exports, and turnover largely disregards climate mitigation benefits from the development and deployment of LCET. This disregard, in turn, diminishes the chances to receive funding for LCET compared to other sectors in the competition for public funds.
- A large emphasis on additionality indicators in the assessment of RD&D support schemes may favour cautious state intervention rather than strong industrial policy push for LCET.
- Disregarding CO_2 emissions as an assessment indicator impacts the selection of LCETs that are worth supporting. If mitigation potential of a technology becomes a secondary concern in the selection of support-worthy RD&D, uncertain but potentially very effective LCET is less likely to receive funding, even though it might be needed for reaching the Paris Targets.

4.3. Internalising the climate externality: MEPS and carbon pricing

Background

The motivation of Paper 3 was to better understand how MEPS could be used to realise radical energy efficiency improvements, which are needed to reach the Paris Targets, and to confront the effectiveness of MEPS with that of carbon pricing. For this purpose, LCCs for four home appliance types on the UK market were modelled considering a cost for emitting CO_2. Appliances included tumble dryers, dishwashers, refrigerators and televisions. Firstly, a significant SCC was introduced in an LCC optimisation model in order to find out by how much standards need to be tightened to account for this climate externality. Secondly, one and two efficiency classes more stringent MEPS were modelled to investigate the switching prices needed to incentivise a comparable shift with carbon pricing.

Conceptual framework

At the core of this paper was the analysis of policy *outputs* (energy efficiency standards and carbon prices) and their design features. The paper considered both market failures and behavioural insights in the analysis of *mechanisms* that influence the *outcomes* of MEPS and carbon pricing respectively. On the one hand, it covered how carbon pricing, the central policy intervention within the green growth framework, can internalise the climate externality (market failure). On the other hand, behavioural insights about people's purchasing decisions in response to changes in LCC of appliances were considered in the model via (implicit) consumer discount rates, which reflect myopia, reference-dependent preferences and bounded rationality.

Key results

The analysis in Paper 3 showed that introducing SCC into LCC models did not drastically change the order of LCC of energy efficiency classes and in most cases the efficiency class with the lowest LCC did not change (see Figure 10 for the examples of tumble dryers and televisions). More detailed modelling revealed that an SCC of USD 150 per tonne of CO_2 would result in a shift of the LCC optimum for tumble dryers by 25 kWh (7%) of annual energy consumption.

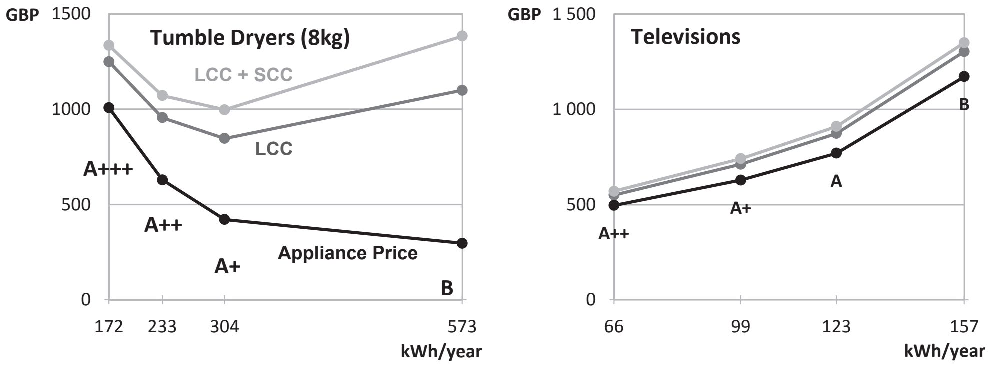

Figure 10
LCC curves for appliances in the UK, with and without SCC.

This in turn suggests that progressive MEPS (e.g. moving up one or more energy classes), can easily internalise the climate externality. For televisions, on the other hand, rational consumers should already be incentivised by electricity prices (no matter whether they include a carbon price or not) to purchase the models with the lowest LCC, and thereby abate emissions.

When looking at the carbon prices required to switch to an average tumble dryer in a higher efficiency class (see central values in Figure 11), it becomes clear that these 'switching prices' are much higher than any carbon pricing scheme in operation right now. This finding holds even if assumptions about the emission factor of the electricity grid or the electricity price are varied.

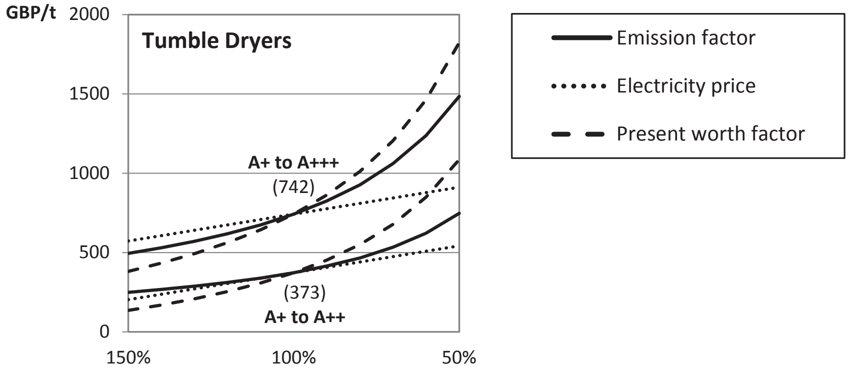

Figure 11
Sensitivity analysis of switching prices for energy efficient appliances. Displayed carbon prices represent a shift of MEPS by one or two efficiency classes. Sensitivity of results was tested with respect to changes in emission factor (100% = 413 gCO_2/kWh), electricity price (100% = 0.14 GBP/kWh) and present worth factor (100% = 10).

From a behavioural perspective, the effect of a variation in the present worth factor on switching prices is interesting, as it illustrates how a higher (and more realistic) consumer discount rate (= lower present worth factor) drives up the switching prices even further. Hence, the results suggest that progressive MEPS can lead to energy efficiency improvements that would be hard, if not impossible, to incentivise with carbon pricing policies.

In the context of RQ 4, these results of Paper 3 illustrate how behavioural anomalies, expressed in high consumer discount rates, lower the incentive effect of carbon pricing interventions for purchasing energy efficient appliances.

Policy implications

- Much more stringent MEPS are able, in principle, to achieve a mitigation effect that is consistent with the 1.5°C target.
- The results from LCC modelling of four home appliances show that a significant climate externality can be captured by MEPS that are not much more stringent than current levels.
- MEPS guarantee behaviour change. Unlike carbon pricing, which do not mandate energy efficiency but only incentivise it, they are not affected by behavioural anomalies or market failures that reduce effectiveness.

4.4. Framing effects in the valuation of climate change mitigation

Background

Paper 4 presents a comparison between different payment mechanisms for the CO_2 emissions of individuals in Sweden. By investigating policy-relevant differences in WTP between these mechanisms, the so-called payment vehicles (PV), the paper aimed to support the design of carbon pricing mechanisms. Moreover, the paper addressed the question whether a uniform carbon price (explicit or implicit) is justified, considering both the urgent need for climate change mitigation and potential differences in WTP between carbon pricing instruments.

At the core of the contingent valuation survey that was used to collect data was the randomised use of four different PVs for the same respondents (n=500). For each PV the size and range of bids presented to respondents was the same (see grey area in Figure 12). Effects of PV choice were thereby isolated from effects driven by differences in study context and sampling. The four PVs researched in this paper were climate surcharges on fuel, on short-distance flights, on long-distance flights and voluntary carbon offsetting (VCO) via the purchase and cancellation of EU ETS allowances.

Conceptual framework

The policy assessment provided in this paper only partly followed the green growth climate policy process outlined in Figure 7, but instead focused on the complementary pathway to drive policy *outcomes* with behavioural insights. The analysed policy *output* was carbon pricing, an economic instrument that fits well into the green growth concept and that has the potential to (partly) internalise the climate externality and lead to climate change mitigation *outcomes*. The behavioural *mechanisms* that were discussed in this paper included the low-cost hypothesis (WTP for environmental impacts higher in situations that imply low costs) and free-riding (people do not like to contribute to public good if they know others can free-ride).

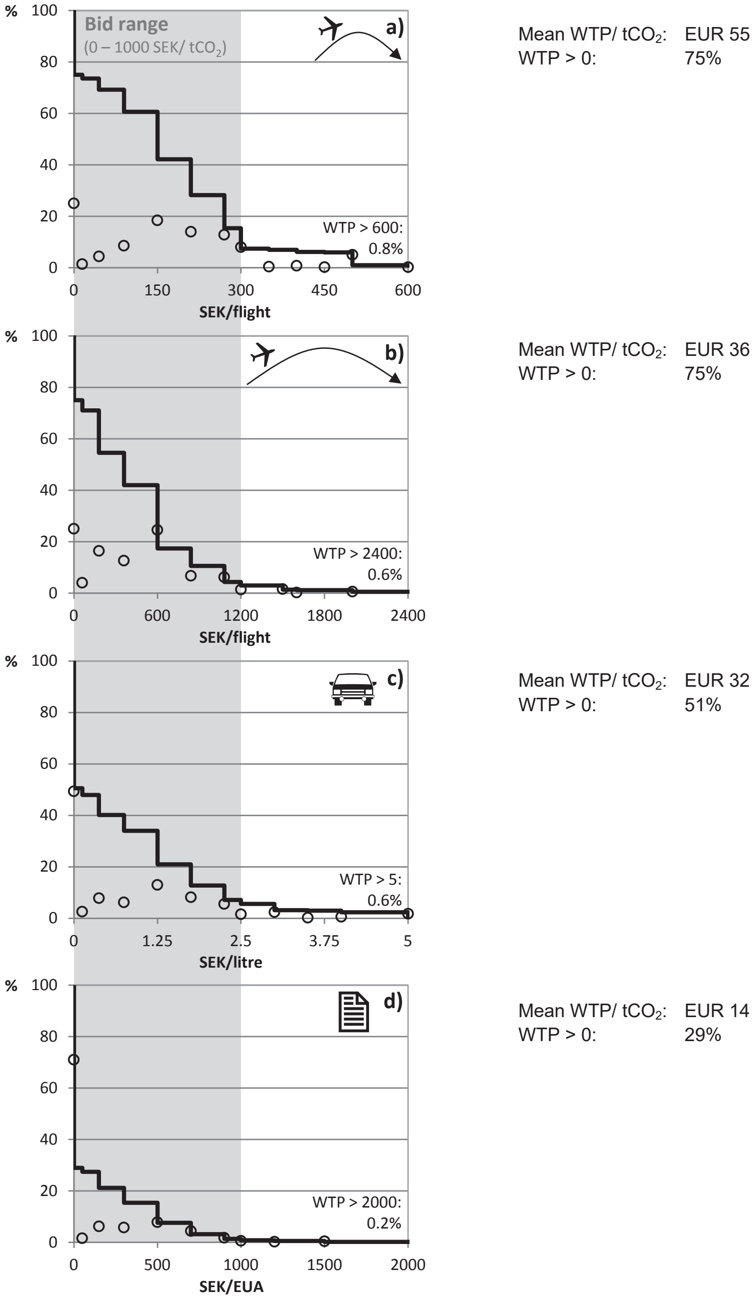

Figure 12
Bid acceptance levels for the four payment vehicles: surcharges on a) short-distance flights, b) long-distance flights, c) motor-vehicle fuels, and d) purchase of EUAs. The circles indicate acceptance at given bid levels; the solid lines show cumulative acceptance. The grey area spans over the survey's bid range. At the time of study SEK 10 were about EUR 1. (Source: Sonnenschein & Mundaca, 2019)

Key results

The main result of the study is that mean WTP values differ between PVs. WTP was highest for a climate surcharge on short distance flights (EUR 55 per tonne CO_2), which was also the PV for which the greatest proportion of people were willing to pay in principle (75%). This was followed by the climate surcharge on long distance flights (EUR 36; 75%), the climate surcharge on fuels (EUR 32; 51%) and voluntary offsetting (EUR 14; 29%). All differences in mean WTP of the possible pairs of PVs were shown to be significant in statistical tests.

The only voluntary PV (offsetting with EU ETS allowances) was associated with the lowest WTP, which supports previous findings that people tend to favour mandatory interventions to instruments that rely on voluntary contributions, and this is often explained by people's aversion to free-riding. Moreover, the study supports the low-cost hypothesis, which claims that in a situation that implies low absolute costs for a respondent, WTP for a good is higher than if absolute implied costs are high. In this study absolute implied costs for the air ticket surcharge on short-distance flights were, by construction, a factor of four lower than for long distance flights (due to the differences in CO_2 emissions), and as predicted by the low-cost hypothesis, the WTP per tonne of CO_2 was significantly higher for short-distance flights.

Finally, the results of this study also illustrate that WTP values are sensitive to changes in the assumed carbon intensity of the respective energy-consuming activities (in particular air travel). The higher the carbon intensity of a flight for which a certain WTP is expressed, the lower is the derived WTP per tonne of CO_2.

The findings of Paper 4 mainly relate to RQ 3, which deals with the factors behind the acceptance of carbon pricing interventions. The acceptance of carbon pricing instruments included as PVs in this paper was driven by the mandatory nature of the intervention and by the size of the implied (absolute) payment. Moreover, the energy use domain and the respective policy context likely drove acceptance.

Policy implications

- People do not appear to value climate change mitigation with one uniform monetary value per each tonne of CO_2, which questions both the suitability and the political acceptability of a uniform explicit carbon price as the main tool to reach the Paris Targets.
- A comparative study of several carbon pricing tools provides orientation for policy (reform) priorities (in this case an air ticket surcharge was clearly favoured over a fuel surcharge or voluntary offsetting).

- As WTP (per tonne of CO_2) appears to be higher for low-cost activities, differentiated carbon prices might well be justified, e.g. different surcharges for long, medium and short-distance flights.
- As WTP depends, among others, on the absolute level of implied costs, more people might be willing to pay if the initial carbon price is low. This price can later be gradually increased.
- Since mandatory payment mechanisms for climate change mitigation are associated with a higher WTP than voluntary mechanisms, stressing that a carbon pricing intervention is mandatory and requires a collective effort may increase acceptability.
- Finally, counting on voluntary offsetting does not appear to be a sufficient climate change mitigation strategy.

4.5. Behavioural insights for a better design of air ticket taxes

Background

Paper 5 is based on the same data set as Paper 4. However, here the focus is not on differences between carbon pricing instruments (the payment vehicles), but only on the climate surcharge on air tickets. The paper investigates the drivers behind people's acceptance of this instrument and their WTP. At the time of the study, an air ticket tax was discussed and planned by the Swedish government, which made the study less hypothetical. The air ticket tax was eventually introduced in 2018.

Conceptual framework

Despite the focus on air ticket taxation, the overall conceptual background of this Paper was the same as for Paper 4. However, the analysis of behavioural *mechanisms* included in addition the attitude behaviour gap (here approximated by a comparing people's acceptance of air ticket taxes with their sense of feeling responsible for reducing their emissions), the response to revenue use options (how does earmarking of revenues affect policy acceptance), and the study of additional drivers behind the acceptance of carbon pricing.

Key results

As already presented in the summary of Paper 4 above, WTP for an air ticket surcharge differs significantly between short and long-distance flights. In a policy context, it is interesting to confront this difference with the differentiated air ticket tax rates that were introduced in Sweden in 2018 (see Figure 13). When looking at WTP and taxes *per flight,* the two perspectives align and, not surprisingly, the values are higher for more expensive and polluting long-distance flights than for short-distance flights. If the perspective is changed and values *per tonne of* CO_2 are considered, the picture changes. WTP is now higher under an air ticket surcharge on short-distance flights, while the tax rate appears to be higher for long-distance flights.

If the policy objective is to reduce CO_2 emissions, the tax-rate per tonne of CO_2 should not differ between short and long-distance flights. Moreover, if WTP values were considered in the ticket tax design, the tax should, if anything, be higher for short-distance flights. At the risk of speculating, these results imply that the Swedish air ticket tax might result in lower emissions reductions (outcomes) than possible, if the differentiated WTP between short and long-distance flights had been considered.

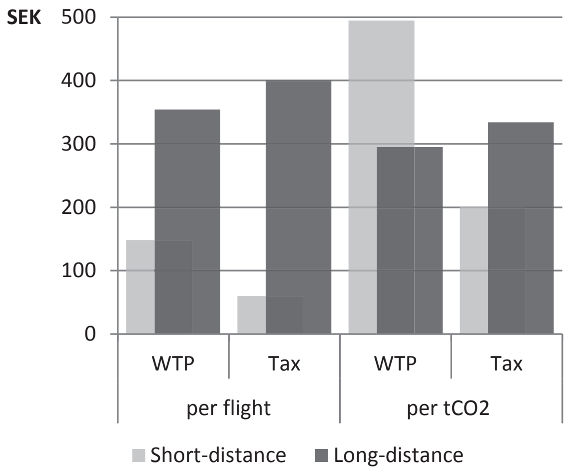

Figure 13
Juxtaposition of average WTP values and Swedish air ticket tax for short and long-distance flights.

Further policy-relevant findings were revealed in the econometric analysis of the variables that help to explain the observed WTP values. Among several socio-demographic variables only respondents' income had a significant and positive effect on WTP, which is a common finding in contingent valuation studies. Moreover, being female significantly increased the likelihood of being willing to pay a surcharge in principle, which supports previous evidence that women tend to behave more environmentally consciously.

Among all significant variables, only being a frequent flyer had a negative impact on the likelihood of being willing to pay and on the WTP amount. All other significant variables tended to increase WTP, which included having a left political view, feeling responsible for your emissions, and being in favour of earmarking revenues from air ticket taxes for climate change related purposes.

The preference for earmarking, which was found to be a significant predictor of WTP, becomes clear in the results of the opinion poll, which revealed strong preferences with respect to revenue use (see Figure 14). Dedicating revenue use to environmentally friendly transport alternatives or directly to mitigating climate change was clearly preferred to using the money for the general budget (the most common revenue use in practice) and even to recycling the money back to the people.

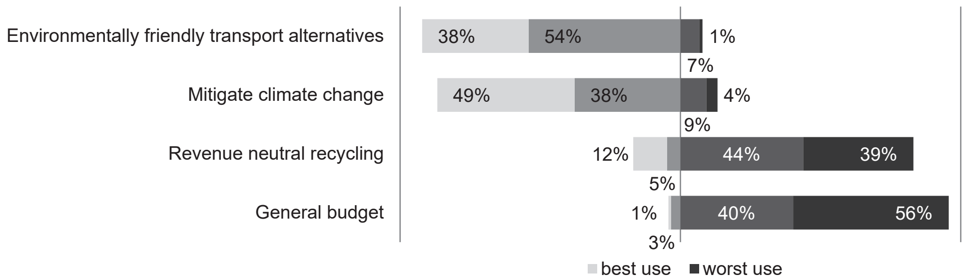

Figure 14
Preferred use of revenues from climate surcharges. Response to the question: “What should the revenues of the climate surcharge on air tickets or fuel be used for?” (n=445). Only preferred revenue use of those 89% of respondents who found that revenue use generally matters is included.

While the share of people who were generally willing to pay a surcharge on air tickets was, at 75%, rather high, there were still some indications of an attitude behaviour gap. The opinion poll that was carried out in addition to the contingent valuation revealed that respondents did not consider air travellers (including themselves) to be responsible for reducing their emission (see Figure 15). This might imply that people were willing to pay more for their flights due to their climate impact, but did not feel responsible to actually limit themselves in their air travel behaviour.

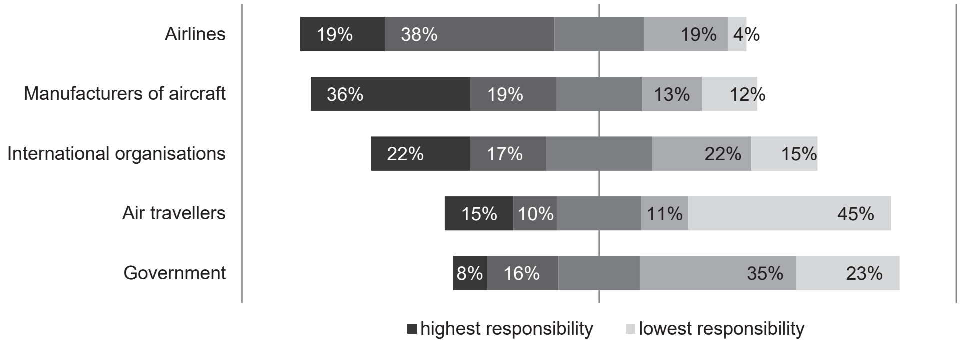

Figure 15
Responsibility for reducing emissions from air travel. Response to the question: "Rank the following actors according to their responsibility for reducing CO2 emissions from private air travel" (n=500).

To sum up, Paper 5 deals with various factors that help to explain the acceptance of carbon pricing interventions (RQ 3) with a focus on air ticket taxation. Being female, having a high income, having a left political view, not being a frequent flyer, feeling responsible for reducing emissions and having a preference for environmental earmarking of carbon pricing revenues were all associated with a higher acceptance of air ticket taxation.

Policy implications

- A mandatory surcharge on air travel emissions appears to be politically feasible as there is a considerable positive WTP for such an intervention.
- If the aim is to steer travel behaviour away from flying, and if distance-specific WTP values are taken into consideration, an air ticket tax (per tCO_2) should be higher for short-distance flights than for long-distance flights.
- Considering that the social cost of carbon is likely more than 100 EUR/ tCO_2, WTP values from this and similar CV studies are lower bound estimates for the value of air travel emissions, indicating that existing carbon pricing policies in the aviation sector are not ambitious enough.
- Earmarking revenues may raise public acceptance and might enable a more ambitious pricing policy.
- The mismatch between frequent flyers' lower WTP and higher impact imply that it is a considerable challenge to implement adequate carbon pricing policies for frequent flyers.

5. Discussion

This chapter aims to synthesise and discuss the results of the thesis horizontally, also taking into consideration (recent) findings from literature. This discussion is organised in accordance with the five research questions (Sections 5.1 to 5.5). The discussion of findings is followed by reflections on the conceptual framework (Section 5.6) and the methodological approach (Section 5.7), including their strengths and their limitations.

5.1. Can green growth policies steer rapid decarbonisation?

The first research questions asked to what extent a green growth strategy is a suitable policy tool for short to mid-term decarbonisation of the economy. Based on the study of Korea presented in Paper 1 the answer should be: to a very small extent. In Korea the partial decarbonisation of the energy system under the country's Green Growth Strategy was not sufficient to offset much of the increase in CO_2 emissions driven by economic growth. Neither absolute nor actual de-coupling could be achieved. This finding is in line with a recent study that investigated de-coupling for the largest economies, in which Korea had the highest economic output elasticity of CO_2 emissions among the advanced economies[13], indicating a very strong coupling between GDP and CO_2 emissions (Cohen et al., 2018).

In other countries that were studied during the research for this thesis, including Sweden (Papers 2, 4 & 5), Germany (Sonnenschein & Hennicke, 2015) and the UK (Paper 3), past developments indicate slightly better outcomes. All three enacted climate change policies over the last two decades and achieved reductions of domestic CO_2 emissions while growing their economies (Cohen et al., 2018; Jiborn, Kander, Kulionis, Nielsen, & Moran, 2018; Le Quéré et al., 2019). However, these emissions reductions are not yet sufficient for reaching the Paris Targets and annual emission reduction rates have to be increased (UNEP, 2018). Moreover, while domestic emissions decreased, consumption-based emissions from imports increased in Sweden

[13] In this study the IMF definition of 'advanced economies' was used, which included among others the US, several EU countries, Japan, Canada and Australia.

(Mundaca, Román, & Cansino, 2015) and the UK (Barrett et al., 2013). For the UK, it could even be shown that the economic output elasticity of CO_2 emissions is only negative for the production-based emissions perspective, indicating de-coupling, while it is still significantly positive for the consumption-based perspective, indicating continuous coupling (Cohen et al., 2018). From a global perspective, even in countries where declining emissions are found due to displacement of fossils fuels and decreases in energy use, the latter is still best explained by lower GPD growth (Le Quéré et al., 2019; Mundaca & Markandya, 2016).

Combined with recent insights from literature, the thesis findings point towards three potential mechanisms that help to explain why green growth policies have not yet had sufficient outcomes. Firstly, they were too focused on economic development rather than climate change mitigation (at an adequate level of stringency). Secondly, they did not consider behavioural insights in the policy design and implementation. Thirdly, they relied too much on economic interventions that were not embedded in a comprehensive policy mix.

Starting with the *focus on economic development*, Paper 1 clearly showed the economic recovery after the Korean (green) fiscal stimulus but no significant impact on emissions. This is not surprising, as parts of the Green Growth Strategy were literally focused on creating new 'growth engines' and less than a third of the spending actually targeted LCET. In addition, Paper 2 showed that economic development indicators (turnover, exports, jobs) played a much larger role in the assessment of public RD&D financing of LCET in the Nordic countries than the climate change mitigation potential of supported technologies. These findings are in line with various studies that criticise green growth policies for their growth focus (Antal & van den Bergh, 2014; Mundaca et al., 2016).

In contrast, the *disregard of behavioural insights* in the design of green growth climate policies has not yet been widely researched. This gap has been noted in the latest IPCC SR1.5 report (de Coninck et al., 2018; Rogelj, Shindell, et al., 2018), in a study that found a focus on economic instruments (Mundaca & Markandya, 2016), and in recent research perspectives that called for better integration of behavioural insights in climate-energy policy research (Steg, 2018; P. Stern, Sovacool, et al., 2016). This thesis addressed this perspective by identifying relevant behavioural insights in the context of carbon pricing and by showing how such insights can improve policy effectiveness (see Sections 5.4 and 5.5 below). In fact, a critical review of large databases with city-level climate mitigation and LCET policies (conducted in the context of this thesis) showed that explicit reference to behavioural insights was an extremely rare exception (Sonnenschein, 2016b)[14].

[14] A comparable finding about the absence of climate change mitigation policies based on behavioural insights was made at the global level (Mundaca, Sonnenschein, Steg, Höhne, & Ürge-Vorsatz, under review).

Finally, this thesis indicates that an *inadequate policy mix* helps to explain the insufficient outcomes of green growth policies. The typical green growth policies alone (i.e. green fiscal stimulus, RD&D subsidies and carbon pricing) do not appear to drive climate change mitigation quickly enough to reach the Paris Targets. A higher ambition level and complementary policies (e.g. regulation and standards) seem to be needed. One key finding of Paper 1 was that Korea's Green Growth Strategy failed largely to deliver because its stimulus package was not complemented by pricing reforms (transport and electricity) or regulation. The inadequacy of single policy instruments was also raised as an issue in Paper 3, which discussed the limitations of carbon pricing in realising the full energy efficiency potential of appliances in the absence of complementary policies (such as MEPS). Furthermore, the results of Papers 4 & 5 suggest that modest carbon pricing alone, while well-accepted, might not lead to the level of behavioural change needed to achieve the Paris Targets. Besides higher tax rates and earmarking of revenues for climate change mitigation, complementary policies might be needed. These findings are supported by IPCC's SR1.5 (Rogelj, Shindell, et al., 2018) and feed into growing research on policy mixes for climate change mitigation and energy transitions (Rogge, Kern, & Howlett, 2017). Specifically, this thesis strengthens the argument that, in order to address behavioural anomalies, carbon pricing (even if carefully designed) is not enough, and complementary policies are needed (Gillingham et al., 2009; Lunn, 2015; Rogelj, Shindell, et al., 2018).

To sum up, the combined results from this thesis and complementary literature imply that green growth policies, as currently implemented, do not drive the short- to mid-term decarbonisation that is needed to reach the Paris Targets. If, however, the focus on economic growth is reduced, if also behavioural insights are taken into consideration, and if an effective and stringent policy mix is implemented, green growth climate mitigation policies have the potential to contribute towards this ambitious goal.

5.2. Are green growth policy objectives in line with the Paris Targets?

> What gets measured gets managed – even when it's pointless to measure and manage it, and even if it harms the purpose of the organisation to do so. (Peter Drucker)

What is the point in assessing whether green growth climate policies are effectively achieving their objectives if these objectives are irrelevant or insufficient to start with? This rhetorical question was addressed by RQ 2, which asked in the context of green growth policies what are the main objectives behind green growth policies targeting LCET and to what extent climate change mitigation is reflected in these objectives. Paper 2 researched RD&D financing instruments for LCET and found that several of these instruments were assessed based on short-term economic performance, that they put great emphasis on additionality of financing, and that they largely disregarded the mitigation potential of supported technologies. It appeared that contributing to climate change mitigation was not their prime objective. Similarly, the main objectives of Korea's Green Growth Strategy (Paper 1) were not limited to climate change mitigation and included additional environmental and social objectives, and the creation of new growth engines. Finally, the discussion of the Swedish air ticket tax in Paper 5 showed that, besides emissions reductions, such taxes also have the objective to raise money for the general budget, which is the case for various carbon pricing mechanisms, including the Swedish CO_2 tax (Hammar & Åkerfeldt, 2011).

These findings question to what extent green growth policy objectives are relevant when confronted with the demanding Paris Targets. It is doubtful to what extent green growth climate mitigation policies can meet multiple objectives (Mundaca et al., 2016; Pahle, Pachauri, & Steinbacher, 2016). The consideration of multiple concurrent objectives, and a subordinate role of climate change mitigation among these objectives, may affect the effectiveness and efficiency of climate policy towards the Paris Targets in at least two ways. Firstly, it may result in the insufficient adoption of LCET and sustainable energy use as policies are not ambitious enough (with respect to decarbonisation) or do not target the technologies or behaviours with the highest potential.

Secondly, mixed (and irrelevant) objectives can lead to an underinvestment in LCET innovation. Paper 2 found evidence that objectives of RD&D subsidies might be influenced by strong concerns about the additionality of public financing, which led to a rather conservative climate financing approach and had a distinct focus on economic development. Such an approach may lead to a situation in which more risky but potentially highly effective mitigation technologies do not receive the amounts of financing that would be beneficial for society, leading to (potential) carbon lock-in effects (Seto et al., 2016). There are two elements of this societal benefit, namely returns on investments and the removal of negative externalities. There are RD&D projects that have an expected return that is above the risk-free market interest rate (see Figure 16), but they still do not receive private financing due to the high risks that are involved (Gaddy, Sivaram, Jones, & Wayman, 2017). It has been argued that the public sector has the opportunity to take greater risks and spread risks better, which – over time – is beneficial from a social welfare perspective (Rodrik, 2014). Public funds are used more efficiently if they support those LCETs that are further away from being commercial (Ek & Söderholm, 2010).

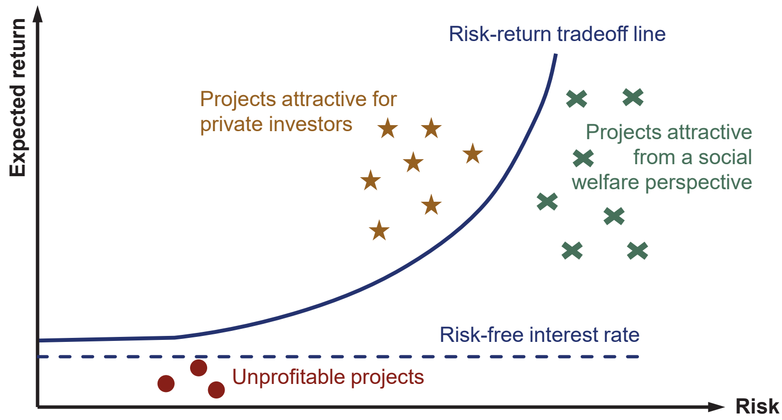

Figure 16
Selection of LCET projects from a private and public investor perspective (illustration based on Grünfeld et al., 2011).

An overly cautious (public) financing approach for LCET overlooks the fact that these investments may remove or avoid negative externalities (via climate change mitigation). While these do not appear on the balance sheets of the supported companies, they nevertheless contribute to social welfare (C. Fischer et al., 2012; Rodrik, 2014). The finding that LCET financing instruments undersupply high risk technologies and ventures is consistent with a study on energy efficiency policy mixes in EU countries, which showed that policy instruments for highly complex and costly (i.e. risky) energy efficiency technologies are lacking (Rosenow, Kern, & Rogge, 2017). Linking back to the quote that opened this sub-section, the risk aversion and profit orientation of public financing instruments might actually be against the 'purpose of the organisation', which in this case is the purpose of public institutions to generate social welfare and steer rapid decarbonisation.

To sum up, the combined evidence from this thesis and new knowledge supports the argument that inadequate policy objectives (e.g. profits and short-term economic development) may result in policy outputs that are too conservative (from a climate mitigation perspective). Even though stakeholders may benefit from irrelevant and mixed policy objectives, they can still lead to poor policy outcomes (e.g. too little development and adoption of LCET). Risk aversion and short-term benefits from the status quo are potential mechanisms that help to explain why climate policy objectives and associated outcomes are not more progressive.

5.3. Can behavioural insights help to increase the acceptance of carbon pricing interventions?

> It is often the short-term acceptability of potential policies, rather than their longer-term efficacy, that determines their scope and deployment. (Kinzig et al., 2013, p. 164)

The third research question of this thesis asked: What are behavioural factors and mechanisms that drive the acceptance of different carbon pricing policies? Policy acceptance was covered in Papers 4 & 5. Before discussing specific implications, it is useful to briefly summarise what are the behavioural factors mediating and the mechanisms influencing acceptance to start with.

In the Swedish study that investigated respondents' WTP for emission reductions under different carbon pricing options, the socio-demographic and psychological factors with a significant influence on acceptance were: being female, having a left political view, not being a frequent flyer, feeling responsible for one's emissions and having a preference for earmarking. Moreover, the study provided indications for relevant behavioural mechanisms (see also Figure 6 above), including conditional cooperation, a negative price elasticity of demand, and the low-cost hypothesis.

However, not all of the identified behavioural factors and mechanisms are equally relevant for policy choice and design. Five policy aspects with high relevance for the acceptance of a carbon pricing intervention were identified based on the evidence of Papers 4 & 5 and complementary literature. These policy aspects include the mandatory nature of carbon pricing, the carbon price level, the use of revenues, the framing of an intervention and its targeting.

To start with, the higher acceptance for *mandatory carbon pricing* found in Paper 4 suggests that carbon pricing should be mandatory. People seem to be willing to pay for their emissions, but they also want to see that others have to pay. This finding accords with several previous studies that found higher acceptance for a mandatory carbon tax on flights than for VCO in Germany (Segerstedt & Grote, 2016), higher WTP for renewable energy under a mandatory scheme than under a voluntary scheme in the US (Wiser, 2007) and higher participation in and contributions to VCO under a collective (and binding) decision rule than under individual choice in an incentivised lab experiment (Löschel, Sturm, & Uehleke, 2017). One exception is a study from the US that showed higher acceptance of paying for VCO on top of an air ticket price than for an equally sized tax, which might be driven by the particularly strong tax aversion among Republicans in the US (Hardisty, Johnson, & Weber, 2010).

Secondly, for mandatory carbon pricing interventions, the influence of *carbon price level* on policy acceptance is well researched and consistent with findings of this thesis. The higher the price is, the lower is the acceptance. Studies about fuel taxation (i.e. implicit carbon taxes) show that once they are established, gradually raising fuel taxes faces less resistance than introducing a significant tax to start with (Flood, Islam, & Sterner, 2010), which provides an additional argument as to why policy-makers prefer weak policy objectives. While low carbon pricing levels might drive acceptance, their downside is that they lower the incentives to invest in LCET innovation and reduce carbon emissions (Fried, 2018; Nordhaus, 2010), and that less revenue is generated that can potentially be used for other climate change mitigation initiatives (Hammar & Sjöström, 2011).

Thirdly, *revenue use* is yet another major design option for carbon pricing instruments. Previous research has shown that environmental earmarking is the most widely accepted option to use revenues from carbon pricing policies (Baranzini & Carattini, 2017; Carattini, Kallbekken, & Orlov, 2019; Drews & van den Bergh, 2016; Kotchen, Turk, & Leiserowitz, 2017; Sælen & Kallbekken, 2011). This was supported by the findings in Paper 5. A potential mechanism for this finding is that people like consistency; they like to see a logical or explicit link between the activity that is taxed and how the revenues are spent (Sælen & Kallbekken, 2011). There are, however, indications that, if a tax is perceived to be already effective in steering behaviour away from environmentally harmful activities, the preference for earmarking is less pronounced (Sælen & Kallbekken, 2011). Whereas earmarking revenues from carbon pricing might enhance acceptance, it is questionable to what extent practical barriers, including the resistance from financial ministries or legal barriers, may prevent this practice.

The issue of revenue use is closely related to the *framing* of carbon pricing policies. It matters to people whether an intervention is presented as a tax (that goes to the general budget) or a surcharge (the revenues of which are used for climate change mitigation). While in this example there actually is a design difference (revenue use), previous studies have shown that for two totally equal instruments the one that is labelled as 'tax' is less accepted than the other (Brannlund & Persson, 2012; Kallbekken, Kroll, & Cherry, 2011). This is one example for framing, which is an umbrella term for various choice architecture approaches. Other examples in the context of carbon and energy pricing include framing VCO as the default (Araña & León, 2013; Löfgren, Martinsson, Hennlock, & Sterner, 2012) and framing feedback about energy use as losses (Bager & Mundaca, 2017).

Insights about behavioural factors and mechanisms can also be used to *target carbon pricing interventions* at certain groups. One example from Paper 5 is the issue of frequent flying. As people generally dislike free-riding and frequent flyers not only contribute disproportionately much to climate change, but also tend to have a lower WTP for the emissions than others, air ticket taxes could be targeted at frequent flyers or be designed in a way that progressively taxes frequent flying. In a recent study from the UK a 'frequent flyer tax' showed more support than rejection and was the highest ranked tax instrument (Kantenbacher, Hanna, Cohen, Miller, & Scarles, 2018).

Finally, insights about the acceptance of carbon pricing policies may even imply that *sometimes implicit carbon pricing should be chosen over explicit carbon pricing*. Salient policies with visible costs (such as explicit carbon pricing) tend to attract strong opposition from interest groups (and also the general public) who believe that the policies are particularly unfavourable to them, whereas less salient policies (such as implicit carbon pricing via regulation) tend to avoid such opposition (Kantenbacher et al., 2018; E. Rhodes, Axsen, & Jaccard, 2014). To address this issue, the thesis examined in Paper 3 whether less salient regulation (MEPS in particular) can still capture the climate externality as well as carbon pricing with its visible costs. The finding that the climate externality can be easily captured by MEPS indicates that less salient, implicit carbon pricing via regulation is a viable alternative to explicit carbon pricing.

To sum up, the combined evidence from this thesis and recent literature supports the view that behavioural insights can increase the acceptance of green growth climate policies and carbon pricing in particular, e.g. by making it mandatory for all, by using revenues for climate change mitigation, by carefully framing interventions, by targeting them at people with a high footprint, or by choosing implicit over explicit carbon pricing.

5.4. Can behavioural insights help to increase the effectiveness of carbon pricing interventions?

While the previous section dealt with behavioural insights to drive policy acceptance, this section deals with behavioural insights and their policy implications in the context of the *effectiveness* of carbon pricing as one important policy to drive green growth. The section thereby addresses the fourth research question of this thesis, which asked 'what are behavioural factors and mechanisms that drive individuals' adoption of LCET and sustainable energy use in response to carbon pricing?'

To start with, the previously mentioned aspect of *carbon price level* seems to have a clear effect on climate change mitigation. The higher the price, the more LCETs and sustainable energy practices are adopted, which in turn may lead to emissions reductions. The long-term (carbon) price elasticity of demand is clearly negative in all areas of energy consumption (Hammar & Sjöström, 2011), and Papers 4 & 5 seem to support this relationship. For small carbon price incentives, the relationship can, however, be questioned as another mechanism seems to kick in, namely crowding-out.

There is evidence of a (modest) *crowding-out effect* of intrinsic environmental behaviour if financial incentives are provided (Beretti, Figuières, & Grolleau, 2013; Brick & Visser, 2010; Bruns & Perino, 2018), and this appears to be partly driven by image motivation (Ariely, Bracha, & Meier, 2009). Crowding-out effects might undermine the effectiveness of carbon pricing. Previous evidence on the issue has, however, shown that moral crowding-out is particularly problematic for weak pricing incentives (Bolderdijk & Steg, 2015; Gneezy, Meier, & Rey-Biel, 2011). This is an argument in favour of higher carbon prices, whereas the previous section showed that high carbon prices can be problematic for acceptance. The complexity of the behavioural response to modest carbon prices illustrates that much more attention needs to be given to the (cognitive or motivational) mechanisms that lead from a given policy to its respective outcomes.

In addition to the overall price level, the *differentiation of carbon prices* between and within consumers and sectors can also be informed by behavioural insights. Paper 4 provides arguments that differentiating carbon prices may increase their effectiveness. Among consumers, WTP for CO_2 emissions reductions varies depending on the area of consumption and depending on the implied absolute costs (Blasch & Farsi, 2014; Diekmann & Preisendörfer, 2003). If there is only one price for carbon and it is low, consumers might not change their behaviour in the areas where they have a high WTP. In contrast, if the price is high, they might oppose carbon pricing in areas with low WTP. Differentiated carbon pricing might address differences in WTP and thereby optimise the (publicly accepted) behavioural leverage. Besides this argument about behavioural leverage, differentiated carbon pricing has been supported by insights from economic modelling that takes into consideration related taxes and additional environmental externalities (Boeters, 2014; Landis, Rausch, & Kosch, 2018). The modelling indicates that if, in addition to explicit carbon pricing interventions, implicit carbon pricing is also considered, implementing a uniform explicit carbon price is not efficient and, moreover, hard to achieve in practice.

For areas where it is not possible to introduce mandatory carbon pricing policies, behavioural insights can be used to inform a *more effective design of VCO and VCO policies*. There appears to be potential to increase the effectiveness of VCO by introducing green default rules[15] (Araña & León, 2013; Bruns, Kantorowicz-Reznichenko, Klement, Luistro Jonsson, & Rahali, 2018; Löfgren et al., 2012), designing incentives for VCO as equal matching grants (rather than rebates) (Kesternich, Löschel, & Römer, 2016) and, where possible, introducing collective decision rules (Löschel et al., 2017; Uehleke & Sturm, 2017). Still, the effect of VCO alone can be expected to be limited. Papers 4 and 5 have shown that VCO is neither done a lot, nor well accepted, nor are people willing to pay large amounts for offsetting their emissions via VCO. While behavioural interventions have shown that increases in the participation in VCO can be achieved, these effects tended to be (statistically) significant but small. This is in line with the findings from a recent RCT of nudges to offset aviation emissions, which found that the effect of nudges was limited and, therefore, argued for industry-level, behavioural economics informed regulation (Tyers, 2018). Limited effect size is not an argument against using behavioural insights but for realistic expectations regarding the impact of behavioural interventions.

Likewise, behavioural insights may also lead to the conclusion that *alternative policy approaches can be more effective than explicit carbon pricing*. Paper 3 illustrated that, other things being equal, consumers do not always buy the appliance with the lowest LCC. Myopia, inattention, self-control problems or bounded rationality might keep them from considering full LCC in their purchase decisions and drive up their implicit discount rates (Schleich, Gassmann, Faure, & Meissner, 2016). Therefore, the effect that carbon-energy pricing policies may have on purchase decisions might be overestimated. This may lead in turn to smaller energy efficiency improvements and associated emissions reductions. If behavioural anomalies are difficult or complex to overcome with carbon pricing, introducing regulation is an alternative. MEPS regulate energy efficiency based on LCC optimisation, so that the consumers no longer even have the choice to purchase appliances with (too) high operating costs. Similarly, Paper 5 showed that a low air ticket tax is unlikely to have a large effect on air travel, so that alternative (or complementary) policies are needed.

To sum up, behavioural insights can help to understand (and address) relevant behavioural mechanisms in the design of green growth climate policies. In particular, behavioural insights have the potential to increase the effectiveness of carbon pricing interventions by informing a certain policy design and implementation, or by suggesting alternative policy instruments in cases where the effect of behavioural interventions alone is not sufficient.

[15] Green defaults may even be ineffective (or less effective) under certain conditions (e.g. psychological reactance among subject participants) so inducing people to make an active choice can be a preferable option (Hedlin & Sunstein, 2016)

5.5. Policies at the intersection of green growth and behavioural insights

The final research question of the thesis aimed at tying together the research on green growth and behavioural insights. With due limitations, it addressed what the interface of green growth policies and behavioural insights is, and what can be learnt from analysing both types of policies.

It is important to acknowledge that, so far, there is no (peer-reviewed) research dealing explicitly with this interface. A GGKP working paper entitled 'Changing Behaviours, Changing Policy - Evidence on Behavioural Insights for Green Growth' (Castro de Hallgren & Root-Bernstein, 2018) provides a conceptual background to behavioural interventions as such and a valuable collection of case studies in the environmental and climate domain. Similar work is presented in the OECD book on 'Tackling Environmental Problems with the Help of Behavioural Insights' (OECD, 2017c). However, neither of the two publications elaborates in depth on how behavioural insights can be used in the specific context of green growth.

In order to discuss green growth-specific behavioural insights, it is important to note that green growth is a normative concept with its own objectives (such as economic growth, reduction of environmental impacts, technology change), whereas the concept of behavioural insights is of a different nature as it does not promote specific policy objectives (except for improving policy assessment and policy-making per se). Moreover, behavioural insights do not imply that certain policy instruments are better, but they may work better under certain conditions, with a certain design and for a certain group of people (OECD, 2017a).

When combining the two concepts of green growth and behavioural insights, some cross-cutting observations based on the research of this thesis can be made. First, it becomes clear that not only technology progress as such matters for climate change mitigation, but adoption and use of LCET and sustainable energy use play a central role. Paper 3 showed that several very efficient appliances are offered on the UK market, some of which have among the lowest LCC. Still they are not necessarily the ones that are sold most, and very inefficient appliances with high LCC are still offered and purchased. Paper 5 illustrated the difficulty of incentivising people to reduce non-sustainable energy use (air travel), as many participants were willing to pay for their emissions but did not feel responsible for reducing them. This finding seems to support recent calls to shift research efforts from climate change awareness and concern to behavioural change and climate mitigation action (Steg, 2018).

There are several reasons why merely putting LCET on the market and creating sustainable energy use options is not sufficient to trigger behavioural change. One reason, which is central under the green growth concept, is the existence of market failures. Green growth policies relying on economic instruments reflect the focus on removing market failures, with carbon pricing to internalise the climate externality being the foremost example. However, as shown in this thesis, putting a price on carbon is not always sufficient to drive the adoption of LCET (Paper 3) and sustainable energy use (Papers 4 & 5). Behavioural factors can significantly influence the functioning of carbon pricing interventions. The apparent neglect of behavioural insights in the context of green growth climate policies strengthens the call for an expansion of the market failure approach (Gillingham et al., 2009; Lunn, 2015).

A first step in this expansion is to also consider behavioural anomalies such as myopia in intertemporal choice or loss aversion (Gillingham et al., 2009). Yet, behavioural anomalies do not account for all behavioural insights about decision-making in the context of LCET adoption and sustainable energy use. Further behavioural insights include behavioural mechanisms that are not anomalies, the choice context (Kunreuther & Weber, 2014; Steg, 2008), and psychological and socio-demographic factors (Abrahamse & Steg, 2011). Accounting for these behavioural insights is not an alternative to the market failure approach to policy-making but an additional factor to consider. As Papers 3–5 indicate, carbon pricing policies that aim at internalising the climate externality (market failure) may fail to achieve this aim if they do not account for the specific energy use context, behavioural mechanisms and additional behavioural factors.

If the market failure approach is expanded and behavioural insights are taken into consideration, this will have implications for green growth policies such as carbon pricing. It is questionable, to say the least, whether global, uniform, explicit carbon pricing is still desirable, let alone feasible. To start with, carbon pricing currently covers only 15% of global GHG emissions and implemented prices centre around 15 EUR/ tCO_2 (World Bank & Ecofys, 2018), which is clearly below what is needed to reach the Paris Targets (Stiglitz et al., 2017). The debate about global uniform carbon pricing appears to be of rather theoretic nature, driven by academic and political beliefs. And even from a theoretical perspective it has been argued that carbon pricing should be embedded in a diversified policy mix to address both market failures and behavioural anomalies (Gillingham et al., 2009; Lehmann & Gawel, 2013). This is supported by evidence from this thesis. In Paper 3 it was shown that unrealistically high explicit carbon prices would be needed to create incentives for boundedly rational consumers to purchase efficient appliances. Moreover, Papers 4 & 5 provided arguments in favour of differentiated carbon prices.

By researching behavioural insights for green growth climate policies, it could be shown that behavioural insights are not limited to interventions that address behavioural anomalies, that is, interventions to improve individuals' utility. In contrast, the studied interventions were mainly targeted at the climate *externality*, that is, interventions to improve social welfare (see also Figure 17 in the next section). As has been argued previously in the context of climate change mitigation, studying policies addressing externalities is quite compatible with dismissing rational choice, because assumptions about the rationality of agents are independent of the occurrence of market failures such as the climate externality (van den Bergh, 2010).

When widening the scope from one green growth policy (carbon pricing) to the overall approach, it is questionable to what extent the addition of behavioural insights to technology-focused green growth climate policies can make a substantial contribution to reaching the Paris Targets. It has been argued that adding behavioural change approaches to green growth policies is insufficient and that a more systemic or holistic research and policy approach is needed (O'Rourke & Lollo, 2015). Beyond individuals, it remains unclear what the specific policy interventions are that facilitate or help to induce such *systemic* change. This goes, however, beyond the scope of this thesis.

Summing up, this thesis identified various policy-relevant aspects when studying behavioural insights for green growth climate policies. While behavioural insights do not challenge green growth objectives per se, they support previous findings that technology change as such is not sufficient for climate change mitigation. Moreover, it is argued that behavioural insights can be used both to expand the market failure policy approach, by addressing behavioural anomalies, and to enrich policies addressing market failures (e.g. the climate externality), by taking into consideration behavioural mechanisms and contextual, socio-demographic and psychological factors.

5.6. Reflections on the conceptual framework

The novelty of this thesis' conceptual approach (as outlined in Chapter 3) lies in the explicit consideration of policy interventions that incorporate behavioural insights in the context of green growth climate mitigation policies. While previous economic research has listed market failures and behavioural anomalies and outlined rough policy interventions (Gillingham et al., 2009), systematic joint consideration both of market failures and behavioural insights in policy assessment has not yet been undertaken. Altogether, introducing behavioural insights in the (stylised) process for green growth policy-making was useful both to enhance the scope of policy assessment and enlarge the toolbox of complementary/alternative policy interventions. Moreover, the conceptual approach managed to capture the research for this thesis in all its width, which was a challenge as the research was carried out in the context of two distinct research projects. However, due to the width of the framework, there are several areas that can be further developed and specified by making use of additional theoretical approaches and enhancing the interdisciplinary approach of future assessments.

Firstly, the framework could be expanded by going beyond the stylised policy process and its focus on policy instruments, and also account for implementing actors and policy context. For the development of LCET innovation (Paper 2), it has been shown that actors and their networks are critical success factors (Söderholm et al., 2019). Contextual factors also turned out to be highly relevant in the cases that were studied for this thesis. Studying green growth policies in Korea (Paper 1), a rapidly developing economy with limited domestic energy resources, likely produces different results than studying for instance Sweden, a country with large resources that industrialised much earlier. Moreover, at the level of behavioural mechanisms (Papers 3–5), contextual influence on policy effectiveness was found, including framing effects and case study-specific factors.

Secondly, the resolution of the framework with respect to behavioural insights could be increased. In contrast to market failures, however, which are well defined and researched in one discipline (economics) and associated with a limited (though contested) set of policies addressing them, behavioural insights build on several disciplines and have various diverse implications for policy choice and design (OECD, 2017a). This makes it difficult to provide a structured and coherent taxonomy of policy relevant insights from behavioural sciences. While several policy-relevant behavioural mechanisms have been identified in this thesis and the wider literature, it is difficult to generalise and conceptualise them as they are typically mediated by a range of contextual, psychological and socio-demographic factors (see also Figure 6 above).

Thirdly, the framework (in particular Figure 7) could clarify better that behavioural insights can also be used to inform green growth policies that address market failures and are not an alternative to the market failure policy approach. Conceptually, this implies that behavioural insights can both be used to target internalities, that is the (future) individual costs and benefits that are not factored into a decision (Allcott, Mullainathan, & Taubinsky, 2014; Herrnstein, Loewenstein, Prelec, & Vaughan, 1993), and externalities such as the unconsidered costs associated with climate change (Oliver, 2015). Examples for policies that are both based on behavioural insights and address the climate externality are MEPS that take into account SCC (Paper 3) and differentiated, well-framed carbon pricing (Papers 4 & 5). The fact that such policies may address both externalities and internalities is rather evident for MEPS, which lead to an increased purchase of appliances with lower LCC. But it has also been shown that there is an internality dividend from instruments such as carbon pricing that primarily address externalities, as they offset distortions from underinvestment in energy efficiency (Allcott, Mullainathan, & Taubinsky, 2014).

A fourth way to expand the framework of this thesis is to differentiate types of green growth climate policies that account for behavioural insights. It has been shown that policies based on behavioural insights are not limited to purely behavioural interventions or nudges, that is approaches that cede the liberty to choose to the individual, but also include more traditional regulation and economic instruments (Bhargava & Loewenstein, 2015; Oliver, 2015). It has been observed (and criticised) that 'interventions within the realm of social and environmental psychology predominantly focus on voluntary behaviour change, rather than changing contextual factors [financial rewards, laws] which may determine households' behavioral decisions' (Abrahamse, Steg, Vlek, & Rothengatter, 2005, p. 274). Opening up the concept of behavioural insights to different types of policies, reaching from soft nudges to restrictive regulation, is supported by findings from this thesis. The research papers provided behavioural insights that suggest it is more effective to choose mandatory carbon pricing over VCO (Paper 4) and – even more liberty-restricting – banning the use of inefficient products instead of making their use more expensive via carbon-energy pricing (Paper 3)[16].

Related to this aspect, the framework of this thesis could also be expanded by making an explicit distinction between policy choice and policy design. Making use of behavioural insights (or not) does not automatically imply a choice between, for instance, green defaults and carbon pricing, a nudge and an economic instrument. Instead, carbon pricing policies as such can be designed in different ways, considering behavioural insights to a larger or smaller extent. Policy assessment should play a key role in making this differentiation clear to policymakers.

Several of the above listed suggestions to expand the framework of this thesis can be captured by plotting policies in a three-dimensional policy space (Oliver, 2015). The three dimensions are made up of the three aspect pairs: liberty-regulation, rational-behavioural and internality-externality (see Figure 17). Starting with the latter, it has been shown that behavioural insights can be used for policies addressing internalities, policies addressing externalities or for both at the same time. Moreover, this thesis has also shown that behavioural insights can provide arguments to restrict liberty by introducing regulation. Finally, the degree to which policies are informed by behavioural insights, and to what extent they assume rational choice may vary. The interventions plotted in Figure 17 have an illustrative purpose and their precise positions should not be over-interpreted.

[16] Note that MEPS that consider the SCC were found to be slightly stricter than MEPS that are based on LCC optimisation; and stricter standards restrict more severely the liberty to choose.

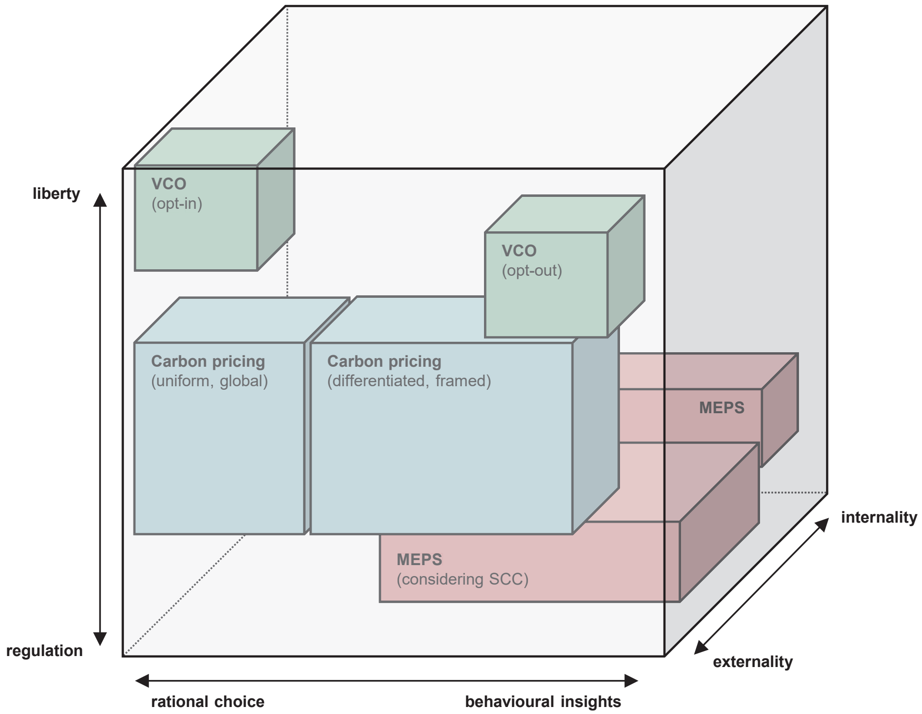

Figure 17
Three-dimensional policy space for climate change mitigation policy (inspired by Oliver, 2015). Policy instruments are differentiated by the degree to which they limit people's liberty to choose, the degree to which they assume rational choice or take into account behavioural insights, and whether they primarily target externalities or internalities.

Summing up, the conceptual framework used in this thesis worked well in capturing the width of the included policy assessments and in highlighting the role of behavioural insights for policy choice and design. Moreover, and informed by complementary literature, the resolution of the framework could be increased in two ways: first, by breaking down the relevant behavioural mechanisms and factors for one specific policy instrument; and second, by placing policy instruments in the three-dimensional policy space presented in Figure 17.

5.7. Reflections on the methodological approach

In this thesis various methodologies were used to assess green growth climate mitigation policy. This methodological diversity addressed among others the limitations of indicator-based ex-post assessment of policy effectiveness. While analysing macroeconomic energy-economy indicators (Paper1) can give a first indication for the effectiveness of green growth policy programmes, this approach neither facilitates the attribution of indicator developments to specific changes in policy, nor does it uncover the mechanisms that drive the outcomes of an intervention (Astbury & Leeuw, 2010). The complementary analysis of several structural factors and higher resolution indicators, as attempted in Paper 1 (see also the extended Kaya Identity in Figure 4), can help to get closer to attribution and increase the validity of findings about policy effectiveness, but it still does not establish causality.

The attribution problem (Scriven, 1991), however, does not negate that green growth climate mitigation policies can and do have an impact. One additional way to research the association between policy interventions and climate change mitigation is to use statistical methods on a larger sample of countries. An exploratory study found, for instance, that there is a significant negative correlation between the Climate Change Performance Index Score[17] of a country and its GDP elasticity of CO_2 emissions (Cohen et al., 2018). Another recent study showed that for countries with decreasing emissions there is a correlation between the number of policies in place and the development of key emission drivers (Le Quéré et al., 2019). While these studies attempt by using statistical methods to show that changes in emissions and their drivers can be attributed to policy interventions, they do not add much explanatory power about the specific mechanisms that lead to the desired outcomes.

A qualitative approach to investigate the mechanisms that connect green growth and climate policy objectives and implementation was explored in Paper 2. The qualitative study of indicators and objectives of RD&D financing policies went beyond the objectives explicitly stated in the documentation of policy instruments. Thereby, the study revealed to some extent what was actually driving policy outputs and not what should drive them ‘on paper’. The study was of an exploratory nature and the methodological approach could in future be expanded by including a larger sample and conducting a complementary quantitative analysis of the performance of the included support instruments.

[17] The Climate Change Performance Index Score evaluates the climate policy performance of 56 countries and the EU (together representing more than 90% of global GHG emissions) and is published on an annual basis by Germanwatch, the NewClimate Institute and the Climate Action Network.

From a bottom-up perspective, the study of behavioural mechanisms and factors influencing the functioning of green growth climate mitigation policies at the individual level can also contribute to the understanding of the mechanisms driving policy effectiveness. For Papers 4 and 5, a survey research method was chosen to study behavioural aspects. Due the hypothetical nature of survey studies (and many lab experiments), using them in policy assessment risks introducing hypothetical bias (Murphy, Allen, Stevens, & Weatherhead, 2005), which has been related to the gap between intentions and action of participants (Ajzen, Brown, & Carvajal, 2004). On the other hand, some of the hypothetical bias can be mitigated by careful study design (Loomis, 2014). Moreover, hypothetical bias appears to be of greater concern for studies interested in quantitative predictions and forecasting than for studies exploring behavioural factors and mechanisms. In order to go beyond the exploration of factors influencing the acceptance and effectiveness of (carbon pricing) policies and get to quantitative behavioural parameters and estimated effect sizes, additional empirical methods are needed, including RCTs and choice experiments (McCollum et al., 2017; Sovacool, Axsen, & Sorrell, 2018).

One of the methodological challenges faced during the research for this thesis was the static time perspective of the methods that were used. In Paper 1 a common problem of policy evaluation was highlighted, namely that time-lags between policy implementation and impacts might not be adequately covered (Crabb & Leroy, 2008). Second, and from a bottom-up perspective, initial effects of behavioural interventions may not be persistent in the longer run (Allcott & Rogers, 2014), and behavioural mechanisms and factors as the ones found in Papers 4 & 5 might change over time. Finally, there is also an evolution of policy instruments. Acceptance of an instrument might quickly change once it is introduced (Schuitema, Steg, & Forward, 2010). Norms and values shift over time and so do the behavioural insights that relate them to policy interventions and outcomes. This echoes calls for more research on the coevolution of social norms and policy instruments (Kinzig et al., 2013).

A more fundamental challenge of the way behavioural insights should be used in policy assessment comes from within economics. It has been argued that many findings from behavioural economics can actually be *integrated* into the utility functions of consumers (Rabin, 2013). Similarly, there are also ways to integrate findings from behavioural economics in cost-benefit analysis (Robinson & Hammitt, 2011; V. K. Smith & Moore, 2010). By integrating behavioural insights into economic analysis, optimisation modelling and the criterion of economic efficiency could keep their role in economic policy assessment. However, scepticism about behavioural welfare economic analysis exists (e.g. about integration of behavioural anomalies) (V. K. Smith & Moore, 2010). Moreover, it is not clear whether this integrated (quantitative) economic analysis would result in different or better policy implications than using behavioural insights directly. In both cases, it is a pre-condition that behavioural insights are robust and have high external validity.

Moreover, integration of behavioural insights in quantitative economic analysis appears to be more relevant for determining economic efficiency than for assessing effectiveness-oriented policies addressing environmental externalities.

In addition to the methods applied in this thesis, there are further approaches to integrate behavioural insights into green growth policy assessment in a deep and rapid decarbonisation context. Two (complementary) research avenues that promise further progress are meta-analyses (based on systematic reviews) and behaviourally realistic modelling (Sovacool et al., 2018). Systematic (literature) reviews are an approach to generate evidence-based information (e.g. for policymakers) with its origin in the medical sciences (Tranfield, Denyer, & Palminder, 2003). More recently, systematic reviews have become more frequent in social sciences (Hansen & Rieper, 2009) and some applications can be found in the context of energy and climate change (e.g. Andor & Fels, 2018; Berrang-Ford, Pearce, & Ford, 2015; Wynes, Nicholas, Zhao, & Donner, 2018). In the field of behavioural mechanisms, systematic reviews can be used to summarise the empirical evidence and, via meta-analyses, come to more robust parameter estimates. There is an ambition to use such parameters to improve the behavioural realism of models (Sovacool et al., 2018), including agent-based models (Janssen & Ostrom, 2006), and integrated assessment models (Lamperti et al., 2018; Safarzyńska, 2018). These efforts are, however, still at the beginning and face many challenges.

Finally, all policy assessment approaches and methods, including the ones of this thesis, face one common challenge: how to generate meaningful results with the limited resources and time-frame available. Considering that comprehensive decarbonisation by mid-century is required to reach the Paris Targets, it is a risky strategy to continue relying on long and resource-intensive ex-post assessment, which might arrive at very similar conclusions from earlier studies. Similarly, slowly piling up more and more empirical evidence about behavioural insights is unlikely to generate results that call for a radical shift of climate mitigation policy. Ambitious (new) policy interventions are needed to challenge incrementalism in climate mitigation policy (Coglianese & D'Ambrosio, 2008), and this should be reflected in policy assessment approaches.

Summing up, it was useful to bring together different methods to study and assess climate change mitigation policy interventions. This multi-method approach enabled the understanding of the diversity of aspects of such interventions, reaching from the role of policy objectives to the role of behavioural mechanisms. In order to get to more robust results and policy advice, methodological limitations need to be overcome and more empirical evidence is needed. This might not be enough, however, as the scale of the climate change challenge calls for radically new approaches.

6. Conclusions

The following sections conclude this thesis by summarising its main insights about policy assessment, choice and design in the context of green growth and rapid decarbonisation (Section 6.1). Moreover, the thesis' main contributions to theory are highlighted (Section 6.2), the overarching policy recommendations are described drawn (Section 6.3), and areas for future research are outlined (Section 6.4).

6.1. Summary of main findings

The findings of this thesis and related literature suggest that, so far, green growth policies have not been effective in driving the rapid decarbonisation that is needed to reach the Paris Targets. This is explained, among others, by the economic growth objectives of many green growth policies, which led to weak stringency levels and, thus, insufficient climate change mitigation outcomes. Other reasons for the failure of green growth policies to drive emissions reductions were the lack of an effective policy mix, strong emphasis on technology markets per se and the neglect of behavioural insights in policy-making.

With respect to behavioural insights, the thesis examined and identified several factors and mechanisms affecting the design and implementation of carbon pricing. By understanding such insights and addressing them in the design of policies, both the acceptance and the effectiveness of carbon pricing can be increased. More specifically, the effectiveness of voluntary carbon offsetting can be increased through careful framing, but there are clear limits to the emissions reduction potential of voluntary measures. Mandatory carbon pricing appears to be the preferred policy. Its effectiveness, however, also depends on the specific design and implementation features, including the price level, differentiation, and revenue use. In other sectors, where carbon price signals are not seen or considered, regulations and strict standards are required.

Moreover, the findings challenge the focus of green growth climate policies on technology change, which – by itself – is not sufficient for rapid decarbonisation. Changes in behaviour are needed, including the adoption of low-carbon energy technologies and sustainable energy use. In the green growth context, behavioural insights can be used to expand the prevailing market failure policy-making approach, by addressing behavioural anomalies. Moreover, green growth policies addressing market failures (e.g. the climate externality) can be enriched by taking into consideration behavioural mechanisms and contextual, socio-demographic and psychological factors.

6.2. Contributions to theory

Besides findings with regards to contents, the thesis also contributes to theoretical aspects of policy assessment. First, and considering that the time to reach the Paris Targets is already running out, the thesis questions the lengthy and resource-intensive ex-post policy evaluation approach, starting from actual implementation of an intervention and covering its outputs, outcomes and impacts. The policies and their stringency that are needed to reach the Paris Targets have virtually no precedent. This calls for creative, forward-looking approaches to policy assessment, including modelling and experimental studies for policy development under limited resources.

Second, and with respect to green growth, the thesis largely takes a critical stance. More specifically, the results of the thesis question the viability of actual de-coupling of CO_2 emissions from GDP, and it is disputable whether economic growth should continue being a central objective of climate change mitigation policies to start with – particularly in industrialised countries. Moreover, the findings illustrate that limiting the conceptual foundations of policy assessment to traditional concepts such as rational choice, economic efficiency and market failures, runs the risk of missing out alternative strategies and opportunities to maximise policy impacts.

The third conceptual finding relates to the use of behavioural insights in policy-making. It was shown that knowledge about behavioural mechanisms can be used in the assessment and design of policies to address environmental externalities such as the costs associated with climate change. In this context, the research and use of behavioural insights is not an alternative to addressing market failures but a complement. At the level of policy instruments, the use of behavioural insights is not limited to explicit carbon pricing policies, but can also be applied to a variety of policy instruments, including implicit carbon pricing.

6.3. Summary of policy recommendations

> Currently, green growth is the only strategy of mainstream economists and policymakers to address climate change. (Antal & van den Bergh, 2014, p. 165)

> [E]conomists should have more to say about public [climate] policy than assigning property rights and adjusting relative prices (Gowdy, 2008, p. 639)

Whereas the first quote may be slightly exaggerated, a focus on economic instruments and green growth-related polices was also detected in the research for this thesis. The second quote then rightly points out that (behavioural) economics and other social sciences have valuable policy advice to add that goes beyond the, at times, simplistic focus on carbon taxes and trading schemes of mainstream economics. The following summary of policy recommendations synthesises the more specific recommendation of the research papers (see Chapter 4). In this synthesis, the Paris Targets are taken as a given benchmark for green growth climate mitigation policies, which inevitably leads to a focus on policy design and policy effectiveness[18].

To start with, the experience with green growth policies so far has shown that – almost tautologically – *objectives of climate change mitigation policies should have a clear and irrevocable focus on rapid decarbonisation*. Technology change, job creation, exports and above all economic growth may all be valid objectives for economic policy and might even be side-effects of climate policy, but should not be confounded with climate change mitigation per se at the level of objectives. Policymakers adopting or supporting the green growth narrative in industrialised countries need to reconsider policies endlessly promoting economic growth.

With respect to technology change, it appears adequate to *provide significant support to low-carbon technology innovations with high decarbonisation potential*, even if they involve high risks. This is not only demanded by the transformational targets of the Paris Agreement, but also makes economic sense as the public sector can more easily spread or reduce investment risks than individual investors. Moreover, additional societal benefits can be gained from reductions of the climate externality.

Carbon pricing, both explicit and implicit, can and should be an important element of green growth climate mitigation policy. Based on this thesis' findings and new knowledge presented in recent literature, carbon pricing should:

- *Be mandatory.* Carbon pricing needs to be mandatory, as the contribution of VCO, even if accompanied by supportive behavioural interventions, is unlikely to make a meaningful contribution to reaching the Paris Targets.

[18] Note that due to the research scope, the following recommendations mainly apply to industrialised countries.

- *Be comprehensive.* The scope of carbon pricing should be as comprehensive as publicly accepted and can be expanded where people are willing to pay but emissions are not yet priced.
- *Create a strong price signal.* People's willingness to pay should not guide absolute carbon pricing levels, as the prices that are needed to steer behaviour towards reaching the Paris Targets are likely much higher than what people are on average willing to pay.
- *Be designed on the basis of behavioural insights.* Differentiating carbon price levels and framing carbon pricing well can drive its acceptance and effectiveness.
- *Consider implicit carbon pricing interventions instead of or in addition to explicit carbon pricing*. The extent to which the effectiveness of explicit carbon pricing can be increased by behavioural insights is limited. Instead, behavioural insights can be used to inform regulations that reflect a high implicit carbon price.
- *Earmark revenues.* Earmarking revenues may raise public acceptance and enable a more ambitious pricing policy. Moreover, earmarked revenues, if used for climate change mitigation, contribute to the effectiveness of carbon pricing.

Beyond the specific instrument of carbon pricing, the findings indicate that the best use of behavioural insights may not be to implement behavioural interventions in isolation, but to argue for a behaviourally-informed policy choice.

Behavioural insights can also be used to *target climate change mitigation policy interventions at specific groups*, reflecting the heterogeneity in the behavioural response to policies. Knowledge about the contextual, socio-demographic and psychological factors driving policy effectiveness can be considered by focusing efforts on groups that are less likely to change behaviour or have a particularly high impact. One such group is, for example, frequent flyers, who are willing to pay less for their emissions while having a higher impact than less frequent or non-flyers.

The findings from this thesis also have implications for the policy-making process as such. In order to address both the challenge posed by the Paris Targets and behavioural insights about low-carbon energy technology adoption and sustainable energy use, it appears to be appropriate to *avoid a narrow focus on economic efficiency and instead put relevance and effectiveness first in the policy-making process*. This also implies a change in the (economic) policy assessment methods that inform policy-making.

6.4. Future research

Due to the width of the scope of this thesis and the urgency of the challenge posed by climate change, there are many topics into which further research is needed. Among these topics, those that are most closely related to the research presented in this thesis are briefly outlined in the following paragraphs.

To start with, there is a need for further empirical data. In the field of green growth, most research is conducted at the country-level and with a focus on de-coupling, while less evidence exists concerning the contribution of individual green growth policy instruments to absolute emissions reductions. Similarly, further empirical evidence about policy-relevant behavioural mechanisms and the mediating contextual, socio-demographic and psychological factors is needed. There is a need for mainstreaming behavioural insights in policy-making, which also requires synthesising the scattered empirical evidence in systematic reviews and meta-analyses. However, both empirical studies and syntheses should focus on 'keystone behaviours' (Castro de Hallgren & Root-Bernstein, 2018), that is behaviours that contribute to or detract most from climate change mitigation. This is critical in order to go beyond incrementalism in both policy assessment and policy-making.

With respect to behavioural insights, comparably little is known about using behavioural insights for the design of traditional policy interventions, including for instance the area of behavioural regulation. Moreover, little is known about the mediating mechanisms that help to explain the causality between a policy intervention and its effect (or the absence of an effect). Moreover, relatively little research has been conducted that goes beyond individual behaviour and accounts for organisational behaviour (Castro de Hallgren & Root-Bernstein, 2018; P. Stern, Janda, et al., 2016), which is needed to generate new insights at the corporate, institutional and sectoral levels (OECD, 2017a).

From a methodological perspective, further advances are needed in the integration of behavioural insights in climate-economy models (Safarzyńska, 2018). One avenue for bringing more behavioural realism into the large climate-economy models is the use of agent-based modelling (Lamperti et al., 2018). Another avenue is to integrate (robust) behavioural parameters in existing models. The literature reveals, however, that parameterisation of behavioural factors for technology choice has been a long-standing challenge for modellers.

Another methodological aspect concerns the research process as such. It has been suggested that, for a transformational research topic such as climate change mitigation, transdisciplinary research approaches are appropriate in order to explore and address real-world problems and contribute to solutions (Lang et al., 2012). Transdisciplinary research can be conducted by including stakeholders of decarbonisation policy directly in the research process (Stock & Burton, 2011), which is an area for further research with respect to green growth climate mitigation policies and behavioural insights.

Finally, there is a particular need for further research concerning policies with the potential for rapid and radical emissions reductions. Many ex-post climate policy assessments have focused on marginal or incremental changes of incentives, economic structure, efficiency or technology, which reflects the scope of the analysed policies. In contrast, mitigation pathways consistent with the Paris Targets require rapid transformational changes, particularly if overshoot of emissions (and related risks) are to be avoided. An interesting area of study is, hence, the evaluation of concrete policy interventions for rapid transitions/transformation. Much research is carried out in the field of socio-technical transition theory (e.g. Bulkeley, Castan Broto, & Edwards, 2015; Geels, 2012; Verbong & Geels, 2010), but this is frequently limited to ex-post analysis and often stops short of analysing (or suggesting) specific interventions that have the potential to drive the transition.

6.5. Concluding reflections

My main reflection on the findings of the thesis and the research process along the way might not sound revolutionary. It is simply that green growth climate policy and its assessment need more *consistency*. To start with, there needs to be consistency between the objectives of national climate policy instruments and the Paris Targets. Green growth policy objectives that make climate change mitigation contingent on economic growth do not contribute to consistency in this respect.

Secondly, there is the inconsistency between objectives and actual policies. Current policies are largely inadequate to reach green growth climate policy objectives (even if these objectives are insufficient to start with). While objectives certainly have their role in the policy process, there often seems to be inconsistency between the scale of the problem and the level of political ambition. Even if successive learning is important for progress, meeting irrelevant objectives undermines the nature of public policy development.

Thirdly, there is frequently inconsistency between the subject (and approach) of policy assessments and the urgency and scale of the decarbonisation challenge (an aspect that could also be brought forward against some of the assessments included in this thesis). The debate (both in academia and policy) has to shift away from marginal adjustments of the existing policy approach and towards higher stringency levels, more radical policy interventions and comprehensive implementation. By assessing, discussing and fine-tuning the current green growth climate policy approach, we run the risk of normalising an approach that is vastly insufficient.

The fourth and final need for consistency is related to policy design. Using revenues from carbon pricing for climate change mitigation appears to be consistent to people, while using them for the general budget (including climate-harmful spending) does not. People seem to have a general preference for consistency in policy-making. This implies, as a side note, that the consistency between people's advice and their own actions is also important. If politicians (and researchers) preach the great transformation but have a very carbon-intensive life- (and work-) style themselves, they risk being accused of hypocrisy and their trustworthiness might be damaged.

So, let's practice what we preach so that our advice is taken seriously. Let's be open to new approaches to policy assessment and design. And let's research and suggest climate change mitigation policies that are truly consistent with the Paris Targets.

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Paper I

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Decarbonization under green growth strategies? The case of South Korea

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ABSTRACT

The win–win opportunities connected to green growth are appealing to academics and policy makers alike, but empirical evaluations about the effectiveness of green growth policies are still scattered. Taking South Korea as case study, which set up a highly ambitious green growth program in 2009, our research casts light on the extent to which the Korean Green Growth Strategy has been effective in decarbonizing the economy. Our methodology combines decomposition analysis and econometrics with a review of energy and climate policies, including related structural changes. On the short term (2008–2012), most of the drivers displayed an enhancing effect on CO_2 emissions from fuel combustion, with GDP per capita being the strongest driver. From a historical perspective (1971–2012), findings reveal that important drivers, such as energy and CO_2 intensity even worsened their effects during the first years under the Green Growth Strategy. Regression statistics revealed that GDP per capita was in fact the driver with the most explanatory power for CO_2 emissions, followed by energy intensity. The Korean policy mix of modest government support to low-carbon energy technologies and a lack of complementary pricing policies did not deliver the targeted emissions reduction, at least in the short-term. Despite recent policy developments, i.e. the introduction of a renewable portfolio standard in 2012 and an emissions trading system in 2015, several key policy challenges for decarbonization remain.

1. Introduction

The 2008–2009 global financial crisis triggered fiscal stimulus packages around the world. While the main purpose of the stimulus was to get economies back on the economic growth path, several environmental organizations, environmental economists, and policy makers saw this crisis as an opportunity to achieve economic recovery with low environmental impact. UNEP pointed out the “unique opportunity presented by the multiple crises and the ensuing global recession” (UNEP, 2009, p. 4). Moreover, it was argued that “a Global Green New Deal, if implemented effectively and swiftly, has the potential to revive the world economy and reduce its vulnerability to repeated fuel and food crises as well as climate-induced risks.” (Barbier, 2010, p. 20). Within this framework, economic stimulus packages were portrayed as a golden opportunity and entry point into a new green economy, with the low-carbon energy technology sector playing a critical role (IEA, 2009). In many countries (e.g. USA, China, South Korea) clean energy was heavily targeted (UNEP and GEI, 2009). While the opportunities connected to green growth strategies are appealing, there are few studies about their actual success in delivering the aspired win–win outcome. The literature regarding the effectiveness of green growth strategies and supportive policies is scattered. This case study of decarbonization in South Korea in a Green Economy context finds that, mainly due to a lack of ambitious supplementary reforms, public spending under a green growth strategy seems insufficient to offset economic growth effects on CO_2 emissions.

The case of South Korea (hereafter Korea) is sticking out in the green growth debate as, together with China, it became the world leader in green growth spending. With 80% the share of green investments in Korea's 2009 economic recovery package of USD 45 billion[1] (representing 3% of GDP) was the largest worldwide (UNEP, 2010). The green stimulus package was already under the impression of President Lee Myung-bak's 2008 announcement of “Low

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[1] When we refer to GDP in the text, we assume an exchange rate of 1100 KRW per USD, which reflects the rate at the time of writing (January 2015) and is close to the average exchange rate over the last five years.

carbon, green growth" as the new development vision for the country. This vision inspired the "National Green Growth Strategy", which was published in 2009. The strategy had "Mitigation of climate change & energy independence" as the first of three objectives. The other two objectives were "Securing new growth engines" and "Improving living standards and enhancing national status", which included only the improvement of water and flood management and the construction of railways as further actions with direct relation to environmental goals (Presidential Commission on Green Growth, 2009). The Green Growth Strategy and its primary focus on climate change mitigation are reflected in several policies, above all the Five Year Plan for Green Growth (2009–2013), which emerged from and overlapped with above mentioned stimulus package, and had a total volume of USD 98.8 billion (OECD, 2012).

There were several reasons for Korea to give a strong push towards the decarbonization of its energy economy. First, Korea is 97% dependent on imports for its primary energy supply (U.S. Energy Information Administration, 2014), which means that energy security and reduced import costs are important co-benefits of climate change mitigation. Second, Korea is an OECD country with consistent and rapid economic growth over several decades (OECD, 2012), but it is one of only three OECD countries that do not have any emissions reduction obligations under the Kyoto Protocol. Third, Korea is a heavily industrialized country with a high share of energy intensive industry, in which a significant part of Korea's economic capacity and welfare is rooted (Jeong and Kim, 2013). Fourth, renewable energy has only a marginal share in both primary energy supply and power generation, which also means that there is no strong domestic market for renewable energy technology, yet (Park et al., 2013). Finally, and most importantly, Korea's CO_2 emissions from fuel combustion increased by 125% from 229 Mt in 1990 to 516 Mt in 2009 (IEA, 2014b).

The Korean commitment towards decarbonization has not only been expressed in the National Green Growth Strategy but also in quantitative targets: Korea committed itself to reducing GHG emissions by 30% till 2020 as compared to a business as usual (BAU) scenario, representing a decrease of 4% compared to 2005 levels. This is the most demanding pledge of any non-Annex I country under the Kyoto Protocol. Furthermore, the First Energy Basic Plan contained targets for the energy intensity of the economy (46% reduction by 2030 as compared to 2006) and renewable energy (increase from 2.4% of total primary energy supply in 2006 to 11% in 2030) (Chung, 2014).

Despite all these relevant drivers and policy commitments, there is a lack of assessment regarding the actual performance of Korea's Green Growth Strategy, in particular from the empirical point of view. Earlier quantitative studies in the context of decarbonizing the Korean energy system have researched: the drivers of CO_2 emission from industry between 1990 and 2009 (Jeong and Kim, 2013), the energy and GHG emissions intensity of 96 economic sectors between 1990 and 2004 (Chung et al., 2009), the role of eco-industrial parks in reducing CO_2 emissions in Korea (Jung et al., 2012), the sector-specific drivers of CO_2 emissions in Korea between 1990 and 2005 (Oh et al., 2010), and the drivers of power sector CO_2 emissions in a scenario analysis for the period 2008–2050 (Park et al., 2013). While these analyses provide valuable quantitative insights about some drivers of energy-related CO_2 emissions, they do not relate their findings to green growth policy programs. On the other hand, recent research on Korean climate and energy policy is scattered. Duffield (2014) provides a qualitative analysis of Korea's first National Energy Plan without putting much stress on its environmental effectiveness. The only explicit attempt we found in the literature is the report "Korea's Green Growth based on OECD Green Growth Indicators" by Statistics Korea. The report provides an interesting summary of several green growth statistics, but neither analyzes these statistics nor assesses the impact of green growth policy on the included indicators (Statistics Korea, 2012). The lack of evaluations of green growth policy programs is likely to explain why there is a discrepancy between the political optimism about the win–win potential of green growth policies on one side, and academic skepticism about the environmental effectiveness of green growth policies on the other side (cf. Antal and Van Den Bergh, 2014; Brahmbhatt, 2014).

Given the lack of knowledge, our research aims to cast light on the extent to which the Korean Green Growth Strategy has been a suitable policy tool for short to mid-term decarbonization of the economy. Our analysis quantitatively unravels key drivers and identifies the extent to which policy efforts have, or not, facilitated decarbonization. The paper combines decomposition analysis and econometrics with a review of energy and climate change mitigation policies; including related structural changes.

The analysis is undertaken in two steps. We first take the Korean National Green Growth Strategy (2009–2013) as a point of departure to analyze recent (2008 onwards) policy efforts to reduce CO_2 emissions. We do this by carrying out an additive decomposition analysis that attributes CO_2 emissions to various drivers, since the indicator CO_2 emissions alone does not have enough resolution to unveil the dynamics that were potentially triggered by policy intervention (methodological details in the next section). Second, and building upon the decomposition approach, we take a longer-term perspective by analyzing Korea's CO_2 emissions using an econometric model with time series data from 1971 to 2012. Questions that guided our analysis included: What have been the most significant drivers of CO_2 emission levels in the short and long term? Which policies (if any) have facilitated the decarbonization of the economy? What can be said about the environmental effectiveness of Korea's Green Growth Strategy? Is Korea on track to reach its 2020 emissions reduction target? And finally, are economic growth and decarbonization compatible? As a whole, our research aims to learn from Korea's experience with using green growth policies to encourage a low-carbon energy system.

The paper is structured as follows. Section 2 outlines the methodology of this study. The results from the short-term decomposition analysis are presented and analyzed in Section 3.1. These findings are put into the context of the long-term development of CO_2 emissions drivers, which were analyzed with econometric tools (Section 3.2). The findings from both parts of the analysis are discussed in the context of structural changes of the Korean economy and its energy system in Section 3.3. Key policy aspects are further analyzed in Section 3.4. Section 4 summarizes implications of our analysis for short to mid-term decarbonization policies. Conclusions are drawn in Section 5.

2. Methodology

The methodology is based on a top-down empirical approach. Building upon the Kaya Identity (Kaya, 1990), our research deploys two complementary analytical tools, namely additive decomposition analysis and an econometric assessment. This study gives emphasis on *environmental effectiveness*, which is primarily assessed by analyzing CO_2 emissions from fuel combustion.

2.1. Decomposition analysis

Decomposition analysis is a useful tool to further the understanding of interactions between CO_2 emissions and socio–economic activities. This understanding can be used as the basis for policies that address the most relevant drivers of CO_2 emissions (IEA, 2014a). The Kaya Identity is a macroeconomic decomposition

Table 1
Parameters and data sources for both decomposition and econometric analysis (full data in Appendix A).

Parameter	Definition	Data source
C	Emissions from fuel combustion (in $MtCO_2$), excluding emissions from marine and aviation bunkers, following the IPCC Sectoral Approach	(IEA, 2014b)
TPES	Total primary energy supply = production + imports − exports − international marine bunkers − international aviation bunkers ± stock changes (in Mtoe)	(IEA, 2014b)
TFC	Total final consumption of energy = sum of consumption by the different end-use sectors, excluding international marine and aviation bunkers (in Mtoe)	(IEA, 2014b)
GDP	Total annual output adjusted by purchasing power parities (ppp) (valued in billion 2005 US$)	(OECD, 2014b)
Pop	All residents regardless of legal status or citizenship, midyear (in millions)	(Statistics Korea, 2014)

equation for energy-economy-environment indicators that quantitatively estimate CO_2 emission levels (Kaya, 1990). The equation typically reads as follows:

$$C = \ Pop{*}GDPpc{*}E\_int{*}C\_int \tag{1}$$

where *C* represents the level of CO_2 emissions from fuel combustion and industrial processes. *C* is the product of four driving factors: *Pop* is population, *GDPpc* is the per-capita GDP, *E_int* is the energy supply intensity of GDP, and *C_int* is the CO_2 intensity of total primary energy supply (*TPES*) (see Table 1 for definitions of indicators and data sources).

Taking the Kaya Identity as point of departure, we decompose CO_2 emissions based on the Logarithmic Mean Divisia Index (LMDI). The advantages of the LMDI method are the ease of using it, the achievement of complete decomposition without residual, the option to carry out both additive and multiplicative decomposition, and the applicability for short time series (Su and Ang, 2012). The LMDI additive decomposition starts off from the basic Kaya Identity:

$$\Delta C = C^T - C^0 = \Delta C_{Pop} + \Delta C_{GDPpc} + \Delta C_{E\_int} + \Delta C_{C\_int} \tag{2}$$

where C^0 are CO_2 emissions from fuel combustion in the base year and C^T are CO_2 emissions *T* years later. The change in CO_2 emissions (ΔC) is split into the respective effects of changes in population (ΔC_{Pop}), economic activity (ΔC_{GDPpc}), energy intensity (ΔC_{E_int}) and carbon intensity of energy (ΔC_{C_int}).

Eq. (2) is further disaggregated into Eq. (3) by separating the transformation effect from the energy intensity effect and by separating the energy mix effect from the carbon intensity of energy effect. This results in:

$$\begin{aligned}\Delta C = C^T - C^0 &= \Delta C_{Pop} + \Delta C_{GDPpc} + \Delta C_{E\_int\_fc} + \Delta C_{E\_transf} \\ &\quad + \Delta C_{E\_mix} + \Delta C_{C\_factor}\end{aligned} \tag{3}$$

where ΔC_{E_transf} is the change in CO_2 emissions that can be attributed the energy transformation effect,[2] which is driven by changes in the ratio of TPES and TFC. Accordingly, $\Delta C_{E\_int\_fc}$ is now based on the TFC of energy. ΔC_{E_mix} refers to the changes in CO_2 emissions driven by the composition of the energy mix, and ΔC_{C_factor} reflects changes in the respective implied emission factors of oil, coal and natural gas. These changes occur as for this analysis *implied* emission factors are used which are not based on the specific carbon content of a fuel. They reflect the ratio between total CO_2 emissions from combustion and the TPES of that fuel. The LMDI formulae for the individual drivers in the additive decomposition Eqs. (2) and (3) are presented in Table 2. The index *i* stands for the different fuel types, such as oil, coal, natural gas and non-carbon energy.

[2] The name "energy transformation effect" is slightly misleading as it merely reflects the ratio between two different metrics of capturing the economy-wide energy, namely TPES and TFC. Between supply and final consumption some transformation takes place (e.g. in power generation), while for other energy products like transportation fuels no transformation happens.

2.2. Econometric assessment

Building upon the Kaya Identity represented by Eq. (1) we defined an econometric model in order to analyze the statistical relationship between key aggregate Green Energy Economy (GEE) determinants for Korea.

$$Y_t = \beta_0 + \beta_1 X_{1t} + \beta_2 X_{2t} + \beta_3 X_{3t} + \beta_4 X_{4t} + \mu_t \tag{4}$$

where Y_t = CO_2 emissions (in million tonnes) from fuel combustion (dependent variable), $t = 1 \ \ldots \ T$ years (=42); β_0 is a constant intercept; β_1, β_2, β_3 and β_4 are the regression coefficients to be estimated for X_1 (*Pop*), X_2 (*GDPpc*), X_3 (*E_int*) and X_4 (*C_int*) respectively; and μ_t is an unobserved error in the model.

Various correlation tests and regression statistics were used for assessing the relationships and contribution of independent variables to historical CO_2 emissions in Korea. First, bivariate correlation tests evaluated the relative degree of 'closeness' (or association) between each pair of variables.

Secondly, partial correlations were calculated to measure the correlation between CO_2 emissions and each independent variable while controlling for the effect of the remaining variables. This step was necessary as more than one variable could convey the same information (i.e. problem of multicollinearity) leading to unreliable estimates and high standard errors. A more important problem is that multicollinearity can make it difficult to draw any inferences about the relative contribution of a particular driver.

Thirdly, using the multiple regression model defined in (2) a stepwise regression analysis quantified the specific contribution of the various drivers of CO_2 emissions. The analysis sequentially assessed the unique impact of each independent variable on CO_2 emissions. If a variable partially explained the behavior of Y (CO_2) it

Table 2
LMDI formulae for various decomposition parameters.

Parameter	LMDI formulae	
Pop	$\Delta C_{Pop} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{P^T}{P^0}\right)$	$P = Pop$
GDPpc	$\Delta C_{GDPpc} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{G^T}{G^0}\right)$	$G = \frac{GDP}{Pop}$
E_int	$\Delta C_{E\_int} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{I^T}{I^0}\right)$	$I = \frac{TPES}{GDP}$
E_int_fc	$\Delta C_{E\_int\_fc} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{U^T}{U^0}\right)$	$U = \frac{TFC}{GDP}$
E_transf	$\Delta C_{E\_int} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{T^T}{T^0}\right)$	$T = \frac{TPES}{TFC}$
C_int	$\Delta C_{C\_int} = \frac{C^T - C^0}{\ln C^T - \ln C^0} \ln\left(\frac{F^T}{F^0}\right)$	$F = \frac{CO_2}{TPES}$
C_factor	$\Delta C_{C\_int} = \sum_i \frac{C_i^T - C_i^0}{\ln C_i^T - \ln C_i^T} \ln\left(\frac{F_i^T}{F_i^0}\right)$	$F = \frac{CO_2}{TPES}$
E_mix	$\Delta C_{E\_mix} = \sum_i \frac{C_i^T - C_i^0}{\ln C_i^T - \ln C_i^T} \ln\left(\frac{M_i^T}{M_i^0}\right)$	$M = \frac{TPES_i}{TPES}$

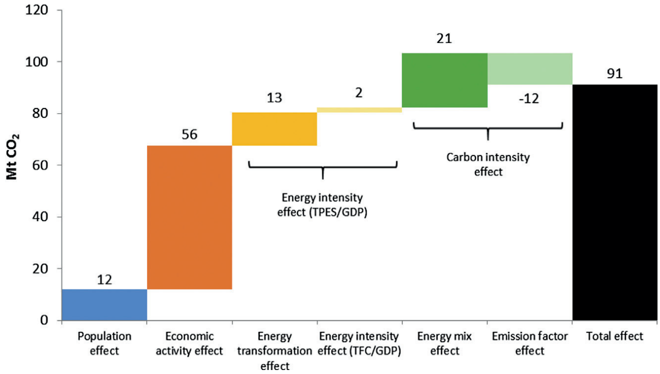

Fig. 1. Results of additive LMDI decomposition of CO_2 emissions from fuel combustion in Korea for the period 2008–2012.

was retained, while all other variables were re-tested to identify whether they were still significant contributors. When a variable no longer contributed significantly to the model, it was removed. This iterative process ran in parallel with multicollinearity tests. The aim was to identify the regression model that explained the greatest part of the variance of CO_2 emissions (i.e. highest adjusted R^2), with p-values below 0.10 (for independent variables), lowest variation coefficients, and no indication of multicollinearity. A variation coefficient $Coef\ Var_j = (Std\ error\ estimate)_j/(Mean\ value\ CO_2)_j$ of the estimated regression model j was calculated in order to evaluate the variability of the dataset and thus the predictive capability (CO_2 variability). A 10% maximum threshold was set (i.e. Coef Var $j < 10\%$). To investigate multicollinearity, Variance Inflation Factors (VIF) were computed to quantify how much the variance of an estimated regression coefficient was increased because of collinearity. A VIF greater than five (i.e., tolerance level below 0.20) was defined a maximum threshold value. That is, any VIF value above five was taken as a strong indication of multicollinearity.

The initial hypothesis was that GDP per capita (g) was most closely correlated with CO_2 emissions, and thus it is an important determinant for explaining the behavior of such emission levels in the country. Unless otherwise stated, all tests and parameters were estimated using a 90% confidence level (i.e. $\alpha = 0.10$).

3. Results and discussion

After the 2008/09 financial crisis, the development of CO_2 emissions in Korea was consistent with the CO_2 rebound effect that was estimated globally. The lowering impact of the crisis on emission levels was "short-lived owing to strong emissions growth in emerging economies, a return to emissions growth in developed economies, and an increase in the fossil-fuel intensity of the world economy." (Peters et al., 2012). Korea is no exception to this and after modest growth – not even reductions – of CO_2 emissions by 11 Mt (2.3%) in 2008 and 14 Mt (2.8%) in 2009, emissions soared up by a staggering 49 Mt (9.5%) in 2010.

While the strong carbon-rebound and the continued growth of CO_2 emissions until 2012 are a first indication for the lack of effectiveness of the Korean green stimulus, further analysis is needed to understand the dynamics of various drivers of CO_2 emissions and whether they have been affected by policies under the Korean Green Growth Strategy. The decomposition of CO_2 emissions from energy between 2008 and 2012 is a first step to understand which factors drove the increase of emissions or mitigated an even further increase.

3.1. Disentangling key drivers for the period 2008–2012

The additive decomposition of CO_2 emissions from energy revealed that between 2008 and 2012 a large share of additional annual emissions was caused by increased *economic activity* (measured in *GDPpc*), which had an emission-enhancing effect of 56 Mt (see Fig. 1). Both the financial crisis and the recovery are covered by the 2008 to 2012 period, in order to avoid distortions of the results by the rebound of GDP and emissions after the crisis.

The rise in emissions caused by the strong economic activity effect was not mitigated by other drivers. On the contrary, changes in the energy intensity and in the energy mix caused a significant *increase* in annual CO_2 emissions, of 15 Mt and 21 Mt respectively.[3] If the energy intensity is based on TFC of energy instead of TPES, the energy intensity effect almost completely disappears. Instead the *energy transformation effect*, which is based on changes in the ratio between TFC and TPES, drives up annual emissions by 13 Mt. This indicates that additional emissions have been triggered by increased losses on the way from energy supply to final consumption, which is due to a higher combined share of coal and natural gas – fuels that are mainly used for power generation, where significant losses occur.

The only mitigating effect of 12 Mt CO_2 occurred because of lowering implied *emission factors* of carbon fuels. This effect can be entirely explained by the decrease in the implied emission factor of oil,[4] i.e. in 2012 less CO_2 was emitted per ton of TPES of oil than in 2008.

The short-term analysis of CO_2 emission drivers does not indicate a win–win outcome of the Korean Green Growth Strategy. Increased economic activity had the expected emission-enhancing effect, but it was not even partly offset by improvements in energy intensity or the decarbonization of its energy mix. The following

[3] Note that the population effect is not further discussed in this paper, as Korean population growth is slowing down and the peak of ca. 52 million is forecasted to be reached in 2030, which is only about 4% more than the current 50 million (Statistics Korea, 2014).

[4] The emission factor effect of oil was −12.4 Mt, of coal 0.9 Mt, of natural gas 0.4 Mt and of other fuels −0.9 Mt. A detailed explanation of the implied emission factor effect of oil follows under the heading "Storing carbon in oil products" in Section 3.3.

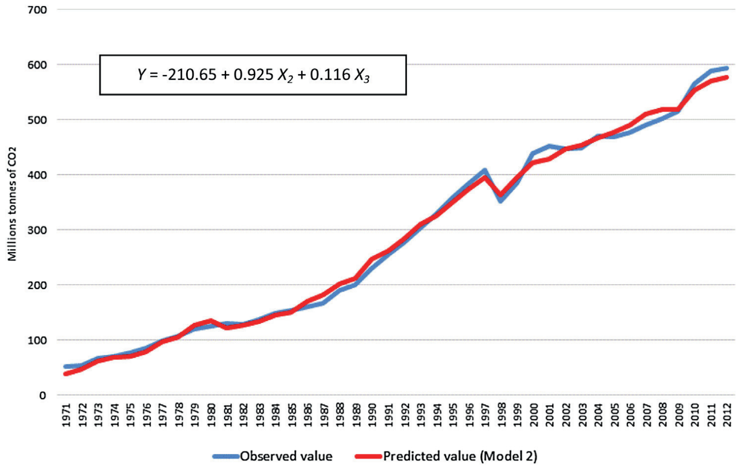

Fig. 2. Observed and predicted CO_2 emissions values from fuel combustion for South Korea (1971–2012).

section further investigates what historically were the main drivers of CO_2 emissions, and whether current developments have been the continuation of (i.e. path dependency) or departure from a long-term trend.

3.2. Unravelling CO_2 emission drivers for the period 1971–2012

The development of historic CO_2 emissions from fossil fuel combustion in Korea from 1971 to 2012 can be best explained by GDP per capita and the energy intensity of the economy. This is the main finding from econometric tests and stepwise regression (details in Appendix B), which resulted in a model where only GDP per capita and energy intensity are left as drivers (see Fig. 2). This model explains 99.6% of the variability of CO_2 emissions.

The findings from stepwise regression analysis are consistent with the results from additive decomposition for the same time period (see Fig. 3, and for more detail Appendix C), where the effect from economic activity (*GDPpc*) on CO_2 emission is clearly dominating. It contributed to increased annual emissions with more than 500 Mt from 1971 to 2012. The energy intensity effect (*E_int_fc*) has mitigated additional CO_2 emissions since the late 90s, whereas the energy transformation effect (*E_transf*) and the energy mix (*E*_mix) effect increased emissions over the same time period. The second mitigating effect besides energy intensity improvements can be attributed to changing implied emission factors (*C_factor*).

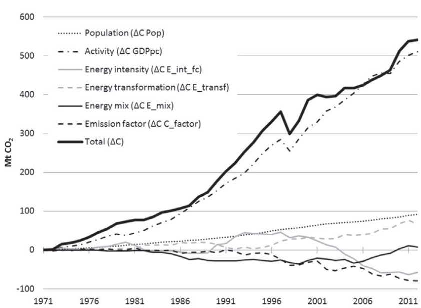

Fig. 3. Results of additive LMDI decomposition analysis of CO_2 emissions (1971–2012).

3.3. Key structural developments

In the following sections, the results from additive decomposition and econometric analysis are put into the context of large structural developments that had an impact on the empirical results for the Korean energy-economy system.

3.3.1. The ‘Miracle on the Han River’

Both in the 2008–2012 and the 1971–2012 time period GDP growth has been the main driver of CO_2 from energy. Per capita GDP consistently grew over the last four decades from USD 2700 in 1971 to USD 8800 in 1990 and USD 21,600 in 2012. The ‘Miracle on the Han River’, a term often used for the economic boom in Korea from the 60s to late 90s, is well reflected in the CO_2 emissions that can be attributed to increased economic activity (as shown in Fig. 3).

The historic development of per capita GDP can be best explained by the rapid industrialization of Korea, which was driven by an active industrial policy and export promotion (Lee et al., 2012), by a high educational standard (Lee, 2012), extensive innovation activity (Chung, 2011), and stable institutions and sound macroeconomic policies (Cho, 2009).

The increasing importance of international markets for Korean economic developments is reflected in the share of value added by exports in GDP, which went up from 53% in 2008 to 57% in 2012, well above the OECD average. At the same time the import-share decreased and Korea developed a large trade surplus (OECD, 2014c).

The two interruptions of economic growth, first during the Asian Crisis in 1998 and then during the Global Financial Crisis in 2008/09, are well-captured by both Model 2 of the econometric analysis and the activity effect in the additive decomposition analysis. The estimated economic rebound, and hence the rebound of the activity effect on CO_2 emissions, was much quicker in the case of the 2008/09 crisis, which had its reason in stable domestic demand, a flexible monetary policy and sound economic institutions (cf. Cho, 2009; Obstfeld et al., 2012).

3.3.2. *Industrialization, tertiarization and industrial restructuring*

The structural change of the Korean economy between 1971 and 2012 had a large impact on energy intensity, and hence CO_2 emissions from energy. It is comprised of three major trends. First, industrialization in the 70s and 80s (continued from the 60s) increased the energy intensity of the economy and therewith CO_2 emissions (see Fig. 4). Second, tertiarization, i.e. the growth of the service sector from 50% value added in GDP in 1980 to more than 60% in the mid-2000s, had a lowering impact on emissions. Tertiarization was mainly driven by growth in producer services, including communication, finance, insurance, real estate, renting of machinery and equipment, advertising and broadcasting (Kim, 2006). Third, the structural change within industry towards less carbon intensive industries, which mitigated additional annual CO_2 emissions of 50 Mt in 2009 as compared to 1999 (Jeong and Kim, 2013).

It is noteworthy that tertiarization has not continued until today. The value added in the services sector as share of GDP reached its all-time high of 61.2% in 2008. After the economic crisis the share of the services sector dropped to 59.4% in 2012, while in the same period the share of industry increased from 36.3% to 38.1% (The World Bank, 2014). This development helps to explain the increase in energy intensity from 238toe per million USD in 2008 to 244toe in 2012, which was a significant driver of CO_2 emissions.

The second factor that influenced the energy intensity effect on CO_2 emissions is the efficiency in the energy system. While not at the core of this study, there are various indications for improved energy efficiency in Korea:

- the efficiency of power generation increased from 33% in 1990 to more than 40% in 2011 (Hussy et al., 2014);
- efficiency improvements in industry mitigated annual CO_2 emissions of about 50 Mt through the 2000s (Jeong and Kim, 2013);
- average CO_2 emissions of cars have decreased from 182 g/km in 2005 to 141 g/km in 2011 (Ko et al., 2014);
- and the TFC of energy in the building sector remained stable between 1990 and 2010 (IEA, 2012), while the number of households, in particular single-occupancy households, rose (OECD, 2014a), and the "total number of house appliances used" went up significantly for most surveyed product categories, e.g. from 1.7 million ACs in 1996 to 13.4 million in 2013 (Statistics Korea, 2014)

However, energy efficiency improvements between 2008 and 2012 where not sufficient to make up for the increase in energy intensity that resulted from the shift towards more energy-intensive economic activity. Hence, the energy intensity effect on annual CO_2 emissions did not display any mitigation in this time period (as shown in Fig. 1).

3.3.3. *From oil to nuclear, to natural gas, to coal and to renewables?*

Historically, changes in the energy mix had varying impacts on CO_2 emissions in Korea. Up until the early 80s, the Korean TPES was dominated by oil for power generation and coal for heating, which had relatively stable shares (see Fig. 5). In the 80s the first nuclear reactors went online and nuclear power reached its all-time highest share in electricity generation of about 50% in 1987 (Choi et al., 2009). This development had a mitigating effect on annual CO_2 emissions.

Throughout the 90s and the early 2000s several changes of the energy mix took place, but their respective impacts on CO_2 emissions largely evened out each other: the share of nuclear in TPES remained stable while natural gas was introduced into the mix and first took shares of coal and later of oil. It is important to consider that implied emission factors of different fuels also changed over time (see section below) so that for instance changes in the share of oil had a different impact on the CO_2 intensity of the energy mix in the early 80s than they would have today.

Starting around the turn of the millennium, the share of coal in TPES, by far the fuel with the highest implied emission factor, rose from about 20% to 30% in 2013. Moreover, no new nuclear reactors were added between 2005 and 2011 (World Nuclear Association, 2015), and existing nuclear power plants generated less electricity, since they underwent additional security checks in the aftermaths of the Fukushima Daiichi accident; including various incidents at domestic nuclear power plants that raised questions about their security (Duffield, 2014).

These trends were not counterbalanced by the modest increase of the share of renewable energy in TPES from 0.5% in TPES in 2005 to 1% in 2013. It is important to note that this modest increase of the *share* of renewables translates into an increase of total renewable energy by 150%, since the TPES increased by 25% over the same

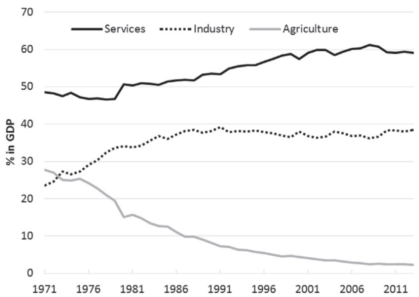

Fig. 4. Value added by sector (The World Bank, 2014).

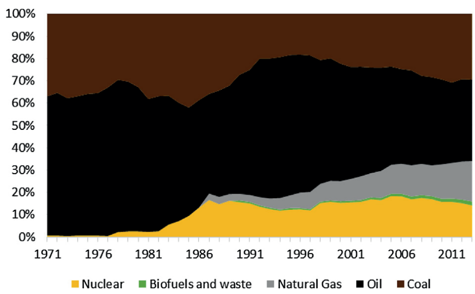

Fig. 5. TPES by fuel 1971–2013.

time period. Despite these positive dynamics, the scale of renewable energy in Korea was too small to affect the CO_2 intensity of the energy mix.

3.3.4. Storing carbon in oil products

Besides changes in the energy mix, carbon intensity of energy was strongly affected by changes of the implied emission factors of different fuels. In particular the per-unit CO_2 emissions from the TPES of oil, which decreased from about 3 tCO_2/toe in the early 80s to less than 2 tCO_2/toe in 2012, influenced the carbon intensity of energy. This development does not reflect changes in the carbon content of oil, but can be explained by the increasing share of the so called "non-energy use" of oil, which is treated as carbon storage.[5] Non-energy use as share of TPES of oil went up consistently from around 10% in the 70s to 38% in 2008 and 44% in 2012.

The improvement in the implied emission factor of oil largely offset the effect from a dirtier energy mix, so that the overall carbon intensity of energy remained stable between 2008 and 2012 at around 2.25 tCO_2/toe.

3.4. The impact of climate and energy policy

The following sections give an overview of the impacts of key Korean climate and energy policies on the development of CO_2 emissions and their drivers.

3.4.1. The green stimulus and Five Year Plan for green growth

Two aspects of the 2009–2012 green stimulus, which was later partly merged into the 2009–2013 Five Year Plan for Green Growth, are relevant for explaining the drivers of CO_2 emissions: the extent to which additional government spending triggered economic growth, and the extent to which this spending had the potential to lower CO_2 emissions.

Korea's share of general government expenditure in GDP is among the lowest of the OECD countries. It slightly grew from 26.6% in 2005 to 30.2% in 2011 and was particularly high in 2009 (33.1%), the year when the Korean fiscal stimulus started (OECD, 2014c). While it is impossible to determine exactly how much of the government spending on green growth programs was *additional* government expenditure that would not have occurred otherwise,[6] it certainly increased spending to some extent. Furthermore, public expenditure triggered growth in the private sector that is not captured by these figures (Hong, 2010). Hence, government expenditure in the context of the Green Growth Strategy caused additional growth of the economic activity effect on CO_2 emissions, even though this effect of additional public spending cannot be quantified.

The envisioned outcome of avoiding additional CO_2 emissions from economic growth by investing in green areas depended heavily on the specific programs that were financed under the Green Growth Strategy. Due to the lack of evaluation of both the stimulus package and the Five Year Plan, it is impossible to determine how much of the spending was directly related to CO_2 emissions. One ex-ante evaluation of the 2009–2012 economic stimulus plan identified 23% of the green spending of USD 38 billion being targeted at the extension of the railway network, 20% at energy efficiency in buildings, 6% at low carbon vehicles, and 6% at low carbon power (Robins et al., 2009; UNEP and GEI, 2009). These figures are similar to the breakdown that the Korean Government provided in its first progress report under the Green Growth Strategy, the only major exception being that the combined share of "green car & clean energy" is reduced from 12% to 5% (Presidential Commission on Green Growth, 2010). It is important to note that just some of the relevant spending – assuming that it was carried out as planned – had the potential for short-term emissions reduction. This includes for example energy efficiency in buildings. On the other hand, investments into rail infrastructure take longer until a potential impact becomes visible. Another example is offshore wind turbines, which take many years from start of construction until grid connection.

Within the Five Year Plan's overall budget of USD 98.8 billion, which included at least parts of the stimulus spending, the shares with relevance for CO_2 from energy were smaller (see Table 3).

Despite uncertainties about the overlap between stimulus and Five Year Plan and the actual implementation of investment plans, a couple of observations can be made. First, the share of public spending with relevance for short-term CO_2 emissions reduction was comparatively small. Second, large infrastructure projects, as the construction of high-speed railways or the Four Major Rivers Restoration Project, made up larger shares of the total spending but lacked the potential for short-term emissions reductions. On the contrary, due to increased demand for resources such as concrete and expanded construction activity, they potentially increased CO_2 emissions. Hence, it is possible that the economic stimulus and the Five Year Plan were short-term drivers of the increase in CO_2 emissions, rather than instruments to mitigate emissions. This is well-reflected in our quantitative analysis, which found that both the energy mix and the energy intensity of the economy worsened their effect on CO_2 emissions between 2008 and 2012. The long-term effects of infrastructure spending under the Five Year Plan are difficult to anticipate in quantitative terms. Since the changes made to the Korean energy system were only marginal, it cannot be expected that the Five Year Plan will trigger large emission reductions in the future.

Moreover, Korea's current Three Year Plan for Economic Innovation (2014–2017) departs from the green growth agenda and puts still more emphasis on economic development. This is reflected in the headline targets of 70% employment, return to annual GDP growth of 4% and more, and increasing GDP per capita into USD 40,000 (Ministry of Strategy and Finance, 2014).

3.4.2. Support to renewables

The core support program for distributed renewable energy has been the One Million Green Homes Scheme, which provides since 2009 financial support for solar PV and solar thermal panels, as well as geothermal energy and small wind power. Furthermore, support at a larger scale was provided to offshore wind projects, tidal energy and wood or pellet fired boilers (IEA, 2012).

Supplementary to these investment subsidies, a government funded feed-in tariff scheme ran between 2002 and 2011. In 2012 this scheme was replaced by a Renewable Portfolio Standard (RPS) under which the largest power generators have to produced or purchase a fixed share of their electricity from renewables, which

[5] It is important to treat this effect as a partly statistical phenomenon, as the category "non-energy use" in the IEA datasets does not necessarily mean that none of this fuel is combusted. If actually less and less of the oil entering the economy is combusted, there would be a clear mitigation effect. If on the other hand, more non-energy use only means, that actual CO_2 emissions vanish from the statistics, the CO_2 statistics show a too positive trend. The IEA lists the non-energy use of fuels as a source of error in their calculation of CO_2 emissions from fuel combustion: "the IEA assumes that 100% of kerosene, white spirit and petroleum coke that is reported as non-energy use in the energy balance is also stored. Country experts calculating the inventories may have more detailed information." (IEA, 2014b).

[6] Furthermore, it is not all clear from the available literature how large the overlaps between the stimulus package and the Five Year Plan were. Some large projects as high-speed rail and restoration of the four major rivers appeared in both plans, which suggests a large overlap (OECD, 2012; Presidential Commission on Green Growth, 2010; Robins et al., 2009).

Table 3
Spending items of the Five Year Plan for Green Growth 2009–2013 with potential relevance for CO_2 emissions from energy (OECD, 2012).

Spending item under the Five Year Plan	In USD bn	Share of total
Total	*98.8*	*100.0%*
Construction of railways	11.7	11.9%
Other spending on climate change mitigation and energy independence	7.8	7.9%
Promoting renewable energy	3.4	3.4%
Nuclear energy development	1.6	1.7%
Developing green villages	0.9	0.9%
Mitigating vehicle emissions	0.5	0.5%
All potential low-carbon energy spending	**25.9**	**26.3%**

started at 2% in 2012 and is going to rise to 10% in 2022 (Duffield, 2014).

The public support to renewable energy was the essential factor for the dynamic growth of renewable energy. It has, however, not been sufficient to help renewable energy gain a significant share in TPES and improve the carbon intensity of the overall energy mix, yet. Furthermore, several policies such as the RPS and subsidies to offshore wind and tidal energy take time to become effective.

3.4.3. Expanding nuclear power

Nuclear power generation appeared in our analysis as one of the few factors improving the carbon intensity of the power mix. The Korean nuclear power program started in the late 50s and built on cooperation with the US. Today Korea has the 6th largest nuclear power capacity in the world and plans further extension (S. Choi et al., 2009). However, it could not keep up with the growth of power demand so that the share of nuclear power generation decreased throughout the 90s and 2000s (as indicated by Fig. 5).

Irrespective, whether nuclear power is regarded a safe and sustainable option, it is clear that short to mid-term decarbonization depends partly on the use of existing nuclear energy capacity. Several scenario analyses for deep decarbonization in Korea go much further and heavily build on nuclear power in the electricity mix (Hong et al., 2014; SDSN and IDDRI, 2014). However, the steep increase of nuclear capacity as foreseen in these scenarios, and to a lesser extent in government plans,[7] is severely challenged by questions surrounding proliferation, safety, costs and, most importantly, the unsolved problem of storing spent nuclear fuels (Duffield, 2014).

3.4.4. Taxing road transport

Fuels for transportation are the energy products that are taxed highest in Korea, at a tax level close to the OECD average (OECD, 2013). While the level of transportation fuel taxes is high compared to other countries, the trend between 2008 and 2012 does not reflect progressive decarbonization policy. The excise duties remained roughly the same, while some fuel prices (before tax) increased. Thus, the share of excise tax in the fuel price decreased in the case of petrol and remained the same in the case of diesel and LPG (see Table 4).

While the 2020 targets for the transportation sector, i.e. GHG emissions reduction by 34% and a fuel efficiency representing 97 g CO_2/km (Duffield, 2014), are ambitious, strong policy instruments are yet to be introduced. The approach of merely setting the regulatory limit of CO_2 emissions to 97 g/km by 2020 is “likely to fall short” to reach a continuous decrease of absolute emissions from passenger vehicles. For that a further reduction of the regulated limit and incentive schemes would be necessary (Ko et al., 2014). In 2009 a motor vehicle tax system that incentivizes the purchase of fuel efficient vehicles was announced for 2015, but later in 2014 it was postponed at least until 2020 (Yonhap News Agency, 2014b).

Considering the increasing CO_2 emissions from road transport, the lack of progressive fuel taxes and CO_2 based motor vehicle taxation illustrate well that green growth programs need supplementary pricing policies and regulation to be effective both on the short and long term.

3.4.5. Market prices for electricity

One of the main policy challenges in improving the energy intensity of the economy was artificially low electricity tariffs. State-owned KEPCO (Korea Electric Power Corporation), which controls more than 90% of power generation operated with annual losses from 2007 until 2012 due to an electricity tariff structure that did not cover costs. In 2013 two electricity tariff hikes were implemented which, in combination with stable fuel costs and a strong Korean currency, resulted in a profit for KEPCO again (Cho and Kim, 2014). Still tariffs were low in comparison to other OECD countries which led to a situation in which the power sector was responsible for a large share of the increase in TPES (Duffield, 2014). Once again, this suggests that a pricing reform to support the green growth spending was lacking, which helps to explain the short-term development of emission drivers that we observed.

3.4.6. Pricing carbon

As long as CO_2 emissions are free or even indirectly subsidized the energy mix cannot be decarbonized. Despite its Green Growth Strategy, Korea heavily subsidized fossil fuel exploration and production. Fossil fuel subsidies based on government tax expenditure totaled USD 4.3 billion in 2011 (Kim, 2013). Ironically, even the Five Year Plan for Green Growth included USD 4.6 billion for the development of foreign oil fields (OECD, 2012).

Table 4
Share of excise taxes in transportation fuel prices between 2008 and 2012 (IEA, 2014c) and change in CO_2 emissions from oil products in road transport.

Tax share in fuel prices (before VAT)	2008	2012
Diesel	32.7%	32.2%
Petrol (95)	48.1%	38.5%
Petrol (92)	51.8%	41.3%
LPG	22.7%	22.1%
CO_2 emissions from oil products in road transport	*77.1 Mt*	*80.4 Mt*

[7] While the First Energy Basic Plan from 2008 envisioned a nuclear share of 41% in the 2030 electricity mix, the more recent Energy Master Plan (2014) decreased this share to 29% in 2035. Interestingly, in both plans the targeted share represents an installed capacity of 43 GW, as the updated Energy Plan assumes a much higher future demand for electricity (Ministry of Trade, Industry and Energy, 2014). In order to realize 43 GW nuclear capacity, the current capacity of 20.7 GW has to be more than doubled. This becomes even more challenging as a couple of the operating power plants will retire in the period until 2035 (World Nuclear Association, 2015).

Table 5
Overview of factors impacting energy CO_2 emissions and related policy factors 2009–2013.

Drivers	Impact on CO_2 emissions*		Driver-related policy challenges
	1971–2012	2008–2012	
Economic activity (GDPpc)	↗	↗	Orienting policies towards improvements in well-being rather than GDP growth[a]
Energy transformation (TPES/TFC)	↗	↗	Introduce carbon pricing to improve the efficiency of power generation (e.g. substitution of coal by natural gas)
Energy intensity (TFC/GDP)	↘	→	Carbon pricing and market pricing of electricity to improve energy efficiency Incentives for energy efficiency in transportation and the residential sector Tax shift from labor to energy in order to incentivize the tertiarization of the economy
Emission factors (CO_2/TPES of various fuels)	↘	↘	
Energy mix (shares of various fuels in TPES)	→	↗	Effective support to renewable energies Clarification of the role of nuclear power Carbon pricing

* The arrows stand for the respective factor's impact on CO_2 emissions (→ = stable; ↗ = enhancing; ↘ = mitigating).
[a] This distinction between the improvement of well-being and GDP growth is informed by relevant literature from ecological economics (cf. Daly et al., 1994; Jackson, 2011), the clear distinction between income and happiness (Easterlin, 1995), and the comprehensive criticism of GDP as the key metrics for economic development (Stiglitz et al., 2009).

Moreover, there were no or only marginal explicit or implicit taxes on carbon in Korea between 2008 and 2012 other than the transportation taxes mentioned above (IEA, 2014c). A nation-wide carbon tax was debated for many years but was never introduced. In the context of the Green Growth Strategy, a target management scheme for CO_2 emissions was implemented – also to establish a basis in monitoring, reporting and verification of emissions and prepare for the later introduction of an emissions trading scheme (ETS). Under the target management scheme and starting in 2012, facilities with emissions higher than 25 kt CO_2 had to agree with the government on CO_2 emission targets and energy conservation targets. From 2014 on also facilities with annual emissions higher than 15 kt were included (GIR, 2014). The impact of the target management scheme on emissions has not been evaluated, yet.

4. Key policy challenges in short and mid-term decarbonization

From an historical perspective, our findings show that the long-term drivers of the steep increase in CO_2 emissions were not halted or even reversed under the Korean Green Growth Strategy (as summarized in Table 5). This is due to the numerous structural and political factors that have kept Korea on an emission growth trajectory: rapid economic growth, the sustained high share of energy intensive industries, the increasing dependence on coal in the power sector, the marginal share of renewable energy, the disputable safety of nuclear power and the challenge of storing spent nuclear fuel, the low retail price of electricity, as well as environmentally harmful subsidies to fossil fuel exploration, production and infrastructure.

In order to identify which policies are necessary to effectively drive a low-carbon economy in Korea, it is useful to have a look at the development of various drivers in two different target-fulfillment scenarios (see Fig. 6). In 'Scenario A' we make the conservative assumptions that TPES will grow 2.1% per year, that GDP (PPP constant USD) will grow 2.6% per year, and that the share of CO_2 emissions from fuel combustion in total GHG emissions will remain constant.[8] 'Scenario B' is a more stringent sustainability scenario, where annual GDP growth amounts to 1% and TPES remains stable. The assumptions about GDP and TPES in Scenario A are optimistic but within the range of government projections, while Scenario B goes far beyond projected developments.

[8] The Korea Energy Demand Outlook from 2014 calculates in its low-growth scenario with an average annual increase of GDP by 2.6% and of TPES by 2.1% between 2013 and 2018 (KEEI, 2014).

In order to reach the 2020 CO_2 emissions target of 30% reduction against BAU the carbon intensity effect has to mitigate 179 Mt CO_2 by 2020 in Scenario A and 101 Mt in Scenario B. The energy intensity effect has to contribute with another 36 Mt (Scenario A) and 46 Mt (Scenario B) respectively.

This brief comparison illustrates that even under the extreme assumptions of 1% annual GDP growth and no increase in TPES, a quick and radical decarbonization of the energy mix is needed to meet the self-imposed climate target. Such a rapid change is unprecedented since 1971 – our initial year of historical analysis. Furthermore, the comparison between the two scenarios illustrates that significantly less decarbonization of the energy mix is needed, if the economy is growing at a lower rate and energy intensity is improving more rapidly.

Despite the start of the ETS in 2015 and the modestly ambitious RPS, full implementation of current policies would only result in emissions reduction to between 630 Mt and 670 Mt CO_2 in 2020 (see range shown in Fig. 7) – falling about 100 Mt CO_2 short of the Korean pledge of 569 Mt (Roelfsema et al., 2014). From a target fulfillment perspective, one example for insufficient policies is the newly introduced ETS.[9] In order to reach the emissions reduction target, the cap will have to decrease to 360 Mt CO_2 by 2020 (Bloomberg et al., 2013). A steep reduction like this is virtually impossible, since for the period 2015–2017 allowances representing 1,687 Mt CO_2 are allocated, on average 562 Mt per year (Cho, 2014), which is still largely above of what is needed by 2020.

In the context of the ETS the RPS can be seen as an instrument to lower the abatement costs for the power sector. It does not affect the cap of the ETS, but compliance for the power sector becomes cheaper. The RPS's 2020 target of 11% renewable electricity is demanding and it will help to drive up the share of renewables on the long run if fully and effectively implemented. Still it will not be sufficient to achieve the decarbonization of the energy mix that is required to reach the emission reduction target of 30% against BAU.

The key policy challenges that can be derived from the ex-post analysis of emission drivers and the ex-ante analysis of two target-fulfillment scenarios are listed below (in brackets we indicate in italics which drivers are addressed):

[9] The ETS started only in 2015. Its first trading period (2015–2017) is a testing phase, in which all allowances are allocated for free (grandfathering). Several last minute changes were made to the ETS, to relax its impact on industry. The cap was increased by about 3% still in September 2014 (Cho, 2014); and in December 2014 the Korean government decided to exempt emissions trading from taxation (Yonhap News Agency, 2014a).

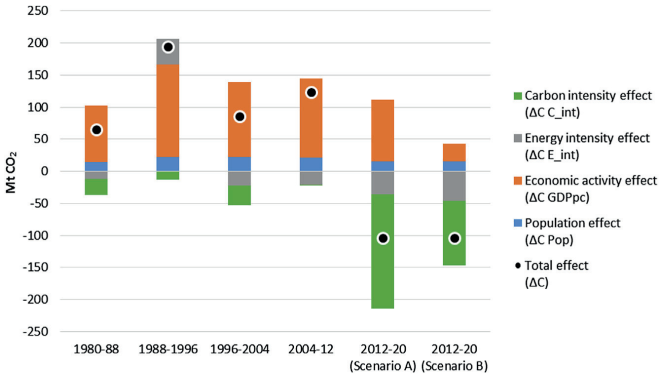

Fig. 6. Additive LMDI decomposition analysis of CO_2 emissions until 2020 based on emissions reduction target and energy intensity target.

- Evaluate the existing portfolio of policy instruments, verify results, withdraw inefficient and ineffective policies and make the necessary corrections so policies are capable of achieving the impacts and outcomes that justify their existence (*all drivers*).
- Find ways to reorient economic policies from GDP growth to improvements of well-being, job creation, and a structural change of the economy (*economic activity*).
- Enable growth in the service sector by lowering labor costs relative to energy costs, e.g. by shifting taxes from labor to energy consumption and CO_2 emissions in a comprehensive ecological tax reform (*energy intensity*).
- Ensure a significant and stable price on carbon to improve the efficiency and carbon intensity of power generation, e.g. by substituting coal with low carbon technologies) (*energy transformation, energy intensity and energy mix*).
- Speed up the transition towards a renewable energy system by ensuring a stable and reliable support scheme both for individuals and for large power generators (*energy mix*).
- Address the challenges of nuclear safety, management of spent fuels and public acceptability and re-evaluate the role that nuclear power can play in the decarbonization of the economy (*energy mix*).
- Increase electricity tariffs by introducing market pricing and taxes in order to manage demand and incentivize energy efficiency (*energy intensity*),
- Reduce CO_2 emissions in road transport by giving incentives for the purchase of low-emission vehicles, e.g. by a revenue neutral feebate system that rewards the purchase of low-emission vehicles and progressively taxes vehicles with high CO_2 emissions (*energy intensity*).

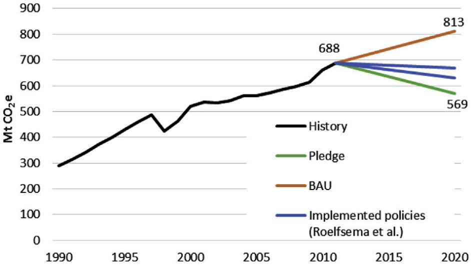

Fig. 7. Different scenarios for the development of GHG emission in South Korea 2011–2020 (adapted from Roelfsema et al., 2014).

Successful decarbonization of the Korean economy needs to address various drivers and cannot rely on an expansion of low-carbon technology alone. The reasons for this are summarized well in the outlook that the International Energy Agency provides (IEA, 2012), stressing that the country is densely populated, heavily reliant on energy-intensive industries, and has not yet started to considerably utilize its renewable energy potential, in particular offshore wind and tidal energy (cf. Kim et al., 2012). Korea is therefore likely to rely on fossil fuels for a large part of its energy demand in the foreseeable future. As energy demand is likely to increase, a reduction in the share of coal and gas might in absolute terms still translate into a rise in consumption. In other words: without a rapid improvement of the energy intensity of the economy (including both energy efficiency and structural change of the economy) CO_2 emissions from energy are likely to rise for another decade and more. The prevailing concentration of large shares of GDP in few energy intensive industries does not only pose environmental risks (Duffield, 2014), but increases the vulnerability of the economy, which means that the strengthening of the service sector represents an opportunity to support a low-carbon economy (cf. Choi et al., 2013; Park and Shin, 2012).

5. Conclusions

Korea's green growth ambitions, and in particular its green stimulus spending, have been frequently referred to as good practice in the international policy arena. Our findings, however, do not fully support this reading of the impacts of the Korean Green Growth Strategy. One key macro-economic indicator of green growth, namely CO_2 emissions from energy, reveals a low performance, i.e. CO_2 emissions increased significantly. We neither observed a change in trends when decomposing CO_2 emissions into various drivers in the short-term (2008–2012), nor when comparing estimated short-term trends to the historical long-term drivers of CO_2 emissions. While it is impossible to attribute driver-specific changes in CO_2 emissions to general policy programs, it is clear that the National Green Growth Strategy of Korea between 2009 and 2013 has not yet been successful in reversing the long-term trend of increasing CO_2 emissions. The targeted peak of emission in 2014 has most likely not occurred, yet.

Some possible explanations for the estimated figures arise from the policy review. First, the specific allocated amount for low-carbon technologies was in fact very modest and measures devoted to short-term effects, such as energy efficiency in buildings and transportation, did not deliver as expected. Secondly, and due to the empirical nature of our study, findings are incapable to capture future long-term effects. In addition, some key policy instruments have been implemented recently: a renewable portfolio standard was introduced in 2012 and the emissions trading scheme was launched in January 2015. Thirdly, the stimulus package was not supported by complementary pricing reforms (transport and electricity) that are also needed to drive a green economy.

The results of our analysis reflect the most challenging aspect of any green or low-carbon growth policy: how to make economic growth truly compatible with low CO_2 emissions, i.e. how to make it coincide with radical improvements of the energy intensity of the economy or the carbon intensity of the energy system. A serious green growth policy program needs to phase out rather than include subsidies to fossil fuels; it further needs to attempt the greening of the existing economy by changing its structure and improving its efficiency instead of merely supporting additional ‘green growth engines’. Above all, it has to go beyond public spending and include ambitious targets and supplementary policies, such as pricing reforms, carbon-energy taxes and stringent regulatory frameworks. Whether all of the above is still compatible with economic growth rates as high as Korea enjoyed them in previous decades is not self-evident. Whereas it is clear that without policies as they are outlined above high economic growth rates do not seem compatible with the decarbonization of the economy.

Acknowledgments

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Appendix

A. Input data for LMDI decomposition analysis and econometric tests

Table 6
Input data for LMDI and econometric tests.

	TPES (in Mtoe)						TFC	CO_2 from fuel combustion (in Mt CO_2)					Population (in million)	GDP (in bn 2005 USD)
	Oil	Coal	Natural Gas	Biofuels & waste	Non-carbon TPES	Total	Total	Oil	Coal	Natural gas	Other	Total		
1971	11	6	0	0	0	17	14	31	21	0	0	52	33	88
1972	12	7	0	0	0	18	14	32	22	0	0	54	34	92
1973	13	8	0	0	0	22	17	40	27	0	0	67	34	103
1974	15	9	0	0	0	23	18	42	29	0	0	71	35	110
1975	16	9	0	0	0	24	19	46	31	0	0	77	35	116
1976	17	10	0	0	0	27	21	52	33	0	0	85	36	129
1977	21	10	0	0	0	32	25	62	36	0	0	98	36	142
1978	24	10	0	0	1	35	27	70	37	0	0	106	37	155
1979	27	12	0	0	1	40	30	78	42	0	0	120	38	165
1980	27	14	0	0	1	41	31	76	48	0	0	124	38	163
1981	24	15	0	0	1	40	31	75	54	0	0	129	39	173
1982	26	16	0	0	1	43	31	74	55	0	0	129	39	186
1983	27	17	0	0	3	47	33	77	60	0	0	137	40	205
1984	27	20	0	0	4	51	36	76	73	0	0	149	40	222
1985	26	22	0	0	5	54	38	73	80	0	0	153	41	237
1986	29	24	0	0	8	61	42	76	84	0	0	160	41	262
1987	29	24	2	0	11	66	45	78	84	4	0	166	42	292
1988	35	25	2	0	11	74	50	94	89	6	0	189	42	323
1989	38	25	2	0	13	79	54	109	85	6	0	200	42	344
1990	50	25	3	1	15	93	65	135	86	6	1	229	43	376
1991	56	25	3	1	15	100	72	158	87	7	2	254	43	411
1992	69	22	4	1	15	111	81	183	82	10	2	277	44	435
1993	78	25	5	1	16	124	89	199	91	12	2	304	44	462
1994	83	26	7	1	16	132	96	215	96	16	2	329	45	502
1995	91	27	8	1	18	145	105	234	102	19	4	359	45	547
1996	97	29	11	1	20	157	112	239	117	26	2	384	46	586
1997	105	32	13	1	20	171	119	248	126	31	2	408	46	613
1998	87	32	12	1	24	156	107	189	130	29	3	351	46	571
1999	95	34	15	1	27	173	118	210	136	36	3	385	47	625
2000	99	42	17	1	29	188	127	220	174	40	5	438	47	678
2001	96	45	19	1	30	191	130	217	186	44	5	452	47	705
2002	97	47	21	2	31	199	135	214	178	49	5	446	48	756
2003	96	49	22	2	34	203	138	212	181	51	6	449	48	777
2004	96	50	25	2	34	208	138	208	195	60	7	470	48	813
2005	92	50	27	2	39	210	140	204	195	64	6	469	48	845
2006	91	53	29	2	39	214	142	196	205	68	7	477	48	889
2007	94	56	31	3	38	222	147	198	211	73	8	490	49	934
2008	90	63	32	3	40	227	147	181	236	75	9	502	49	955
2009	91	65	32	3	39	229	148	182	253	72	9	516	49	959

Table 6 (*continued*)

	TPES (in Mtoe)						TFC	CO_2 from fuel combustion (in Mt CO_2)					Population (in million)	GDP (in bn 2005 USD)
	Oil	Coal	Natural Gas	Biofuels & waste	Non-carbon TPES	Total	Total	Oil	Coal	Natural gas	Other	Total		
2010	95	73	39	3	39	250	158	187	277	91	10	564	49	1019
2011	94	80	42	4	41	260	161	183	298	98	12	590	50	1057
2012	97	77	45	4	40	263	166	184	291	106	12	593	50	1078

B. Detailed results of econometric analysis

First, all independent variables showed the potential to individually explain the behavior of Korea's CO_2 emissions (see). The variable that showed the highest correlation with CO_2 was *GDPpc* (99.4%). However, the fact that independent variables appeared highly correlated indicated early signs of multicollinearity for the regression analysis.

Table 7
Bivariate correlation test (n = 42; all correlations significant at 0.01 level).

	CO_2	*Pop*	*GDPpc*	*E_int*	*C_int*
CO_2	1	0.964*	0.994*	0.660*	−0.912*
Pop		1	0.956*	0.731*	−0.946*
GDPpc			1	0.588*	−0.912*
E_int				1	−0.695*
C_int					1

Estimates from partial correlation tests started to confirm the initial hypothesis: *GDPpc* is the most significantly correlated variable (98.1%) with CO_2 emissions (see). The level of correlation dropped marginally (−1.3%) compared to bivariate correlation tests. This suggested that the relationship between CO_2 and *GDPpc* was slightly mediated by *P, E_int* or *c_int*. Partialling out *P, GDPpc*, and *C_int* individually suggested that *E_int* was the principal mediator (86.8%).[10]

Table 8
Partial correlations tests.

		CO_2
Pop (controlled variables*: E_int, C_int, Pop*)	Correlation	−0.259
	p-value	0.111
GDPpc (controlled variables*: E_int, C_int, Pop*)	Correlation	0.981
	p-value	0.000
E_int (controlled variables*: GDPpc, Pop, C_int*)	Correlation	0.868
	p-value	0.000
C_int (controlled variables*: GDPpc, Pop, C_int*)	Correlation	0.888
	p-value	0.004

Results from the stepwise regression can be summarised as follows (see). First, all variables but *Pop* were introduced in our original model (in the following 'Model 1'), which was based on Eq. (4). Model 1 was significant (F3, 38 = 4710.29; *p*-value = 0.000) and explained 99.7% of the variability of CO_2 emissions (R2 = 0.997). The coefficient of variation for Model 1 (*Coef_VarModel-1* = Std. error estimate (±9.29)/mean value of CO_2 emissions (285.74 $MtCO_2$)) yielded a value of 3.25%, which suggested that large fluctuations of *CO_2* emissions could be explained by the estimated model. However, estimated VIF values for Model 1 revealed strong signs of multicollinearity, with estimated indexes for *GDPpc* and *C_int* in particular, much higher than the defined maximum threshold value.

Table 9
Summary output from stepwise regression analysis.

Regression summary		**R**	**R^2**	**Adjusted R^2**	**Std. Error**
Model 1*		0.999	0.997	0.997	9.29
Model 2**		0.998	0.996	0.996	11.08

ANOVA		**Sum of Squares**	**df**	**Mean Square**	**F**	***p*-value**
Model 1*	Regression	1220304.51	3	406768.17	4710.29	0.000
	Residual	3281.57	38	86.35		
Model 2**	Regression	1218797.91	2	609398.96	4963.60	0.000
	Residual	4788.16	39	122.77		

Coefficients		**β (Standardised)**	**Std. Error**	***t***	***p*-value**	**VIF**
Model 1*	β_0	−403.27	49.63	−8.12	0.000	_
	β_2 (*GDPpc*)	1.000	0.00	48.35	0.000	6.05
	β_3 (*E_int*)	0.140	93.80	11.83	0.000	1.98
	β4 (C_int)	0.097	11536.00	4.17	0.000	7.66
Model 2**	β_0	−210.65	21.88	−9.62	0.000	_
	β_2 (*GDPpc*)	0.925	0.00	74.69	0.000	1.52
	β_3 (*E_int*)	0.116	98.27	9.39	0.000	1.52

*Predictors: (Constant), *GDPpc, E_int, C_int*.
**Predictors: (Constant), *GDPpc, E_int*.

As a consequence, a second simulation round took place. This resulted in 'Model 2' containing only *GDPpc* and *e_int* as significant drivers for South Korea's CO_2 emissions. Model 2 was significant (F2, 39 = 4963.6; *p*-value = 0.000) and the estimated adjusted R2 was still very high: 99.6% of the variability of CO_2 emissions is explained collectively by *GDPpc* and *E_int*. This level of determination was only marginally reduced compared to Model 1 (see Table 6). Although the standard error was slightly higher (±11.08 $MtCO_2$) compared to Model 1, the estimated *Coef_VarModel-2* was equal to 3.87%, which suggested that Model 2 would also be useful in predicting CO_2 emission interval values (i.e. ratio is lower than 10% threshold). VIF measures revealed no signs of multicollinearity, with estimated values for the independent variables equal to 1.52, a value lower than the defined 5 maximum threshold value. Finally, estimated coefficients (standardised) confirmed that *GDPpc* had the strongest impact on CO_2 emission levels.

C. Further details of the LMDI decomposition analysis

The two figures presented below provide a closer look at the drivers energy intensity and carbon intensity of energy in the LMDI decomposition analysis. In most decomposition analyses energy intensity is based on TPES (cf. IEA, 2014b). Fig. 8 shows that basing energy intensity on final consumption, and hence extracting the energy transformation effect, results in a more differentiated picture of the effect of energy intensity on CO_2 emissions. In this case energy intensity based on final consumption had a more consistent mitigation effect on CO_2 emissions since the late 90s than energy intensity based on TPES.

[10] In addition, partial correlation results also revealed that the relationship between *CO_2* and *E_int* was significantly mediated by *Pop* and *C_int* in particular (correlation increased from 66-6%–86.8%). In fact, the relationship between *CO_2* and *Pop* was no longer significant (p-value$_{Pop}$ = 0.111).

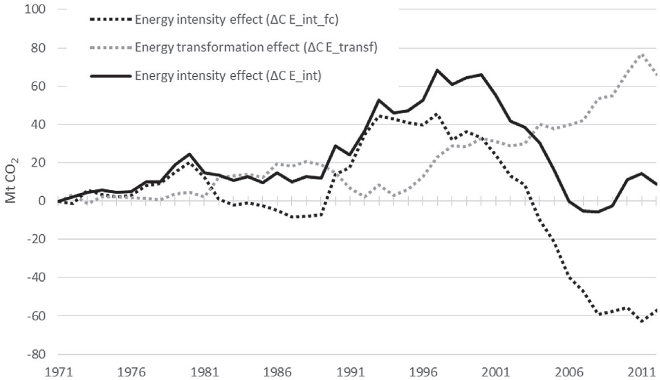

Fig. 8. Results of additive LMDI decomposition analysis of CO_2 emissions – energy intensity and energy transformation effect.

Similarly, splitting the carbon intensity of energy effect into an energy mix and emission factor effect reveals new trends. Both the mitigating effect of lowering emission factors since the early 90s and the increase of emission due to changes in the energy mix after 2005 were "hidden" in the carbon intensity of energy effect.

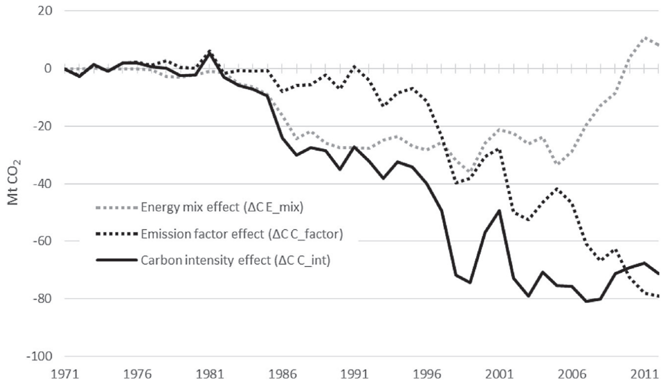

Fig. 9. Results of additive LMDI decomposition analysis of CO_2 emissions – carbon intensity, energy mix and emission factor effect.

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Paper II

The Political Economy of Clean Energy Transitions

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10

Understanding Indicator Choice for the Assessment of RD&D Financing of Low-Carbon Energy Technologies

Lessons from the Nordic Countries

Jonas Sonnenschein

10.1 INTRODUCTION

Many climate scenarios show potential pathways to limit global warming to two degrees as stipulated in the Paris Agreement of 2015 (Edenhofer et al. 2014). These scenarios have in common that in order to decarbonize industrialized economies, further research, development, and demonstration (RD&D) of low-carbon energy technologies (LCET) and of technologies inducing 'negative emissions' are urgently needed (Clarke et al. 2014; Anderson 2015). In addition to new technological solutions, the speed of deployment and the integration of solutions into the energy system are critical factors in climate change mitigation.

When decarbonization scenarios go beyond technological feasibility and economic factors are accounted for, the focus is often on costs and additional investment needs (Gupta et al. 2014). The importance of low abatement costs is well-reflected in some of the main climate policy instruments, such as carbon-energy taxation, carbon trading, and green and white certificate schemes (Somanathan et al. 2014), which induce marginal changes in price structures. From a national perspective, the focus on low-cost abatement is justified as domestic climate change-related benefits do not outweigh the costs of the unilateral adoption of more expensive abatement options (Stavins 2014). Weak (carbon) price signals and—in the case of trading schemes—price fluctuations do not create sufficient incentives to fix the market failures in the generation of LCET change (Jaffe, Newell, and Stavins 2005), leading to underinvestment in RD&D and innovation. Due to spill-overs the social rate of return of RD&D investments is often higher than the commercial return rate (Griliches 1992). Moreover, it is in many cases particularly difficult to finance LCET as it has high capital requirements and a long time to market (Ghosh and Nanda 2010).

In order to scale up RD&D activity in this area, it is critical to know if governmental intervention can correctly identify RD&D initiatives with high social returns that are under-supplied with financing from the market. Thus, it is relevant to understand both the motivation for setting up new public RD&D support instruments and how their success is assessed and measured. As success is a normative concept, different stakeholders may have their own specific criteria or indicators for success of public RD&D in this context.

While there are various methods for evaluating the performance of RD&D support policies, many of them rest on few aggregated indicators, such as public and private RD&D expenditure as well as patent counts (Bozeman and Melkers 1993). These indicators alone do not reflect the complexity and dynamics of public RD&D, let alone innovation processes (Gallagher, Holdren, and Sagar 2006). The quantitative estimation of innovation policy indicators has been frequently criticized for rarely coming to conclusions with high policy relevance (Bergek et al. 2008).

The approach of *evaluating* indicators addresses this criticism without completely abolishing the indictor-based method. Indicator evaluation in the field of LCET RD&D is neither very far developed nor tested. Notable attempts are: Gallagher, Holdren, and Sagar (2006), who discuss the merits of various input, output, and outcome metrics but do not apply a uniform indicator evaluation framework; Wilson et al. (2012: 781), who roughly estimate the suitability of various indicators to research 'directed innovation efforts in response to climate change mitigation'; and Carley, Brown, and Lawrence (2012), who propose an evaluation framework for 'energy-based economic development' which includes the categorization of relevant indicators but not an actual indicator evaluation.

The purpose of this study is to assess the performance of indicator-based evaluation in the context of LCET support policies and to contribute to the structured assessment of potential indicators.

The Nordic countries have been chosen as a geographic area of study. Norway, Sweden, Finland, and Denmark have innovation ecosystems in place which provide dedicated support to LCET. They perform well on indexes related to eco-innovation, such as the Global Green Economy Index 2016 (Dual Citizen LLC 2016), the 2014 Global Cleantech Innovation Index (WWF and Cleantech Group 2014), and the EU Eco-Innovation index (European Commission 2015). However, in Finland and Sweden in particular, the gaps between the evidence of emerging cleantech innovation and the evidence of commercialized cleantech innovation are large (WWF and Cleantech Group 2014). Within cleantech the sub-sector of LCET is particularly challenging due to long times to market and high capital requirements (Ghosh and Nanda 2010). Both the success of their cleantech industry and the remaining challenges in commercialization make the Nordic countries a suitable case study to identify and analyse indicators for the assessment of RD&D support policies.

Section 10.2 in this chapter includes the research design. In Section 10.3 the analysis of the indicator-based evaluation framework is presented. Section 10.4 discusses policy implications of indicator choice and Section 10.5 concludes.

10.2 RESEARCH DESIGN: THE INDICATOR-BASED EVALUATION FRAMEWORK

The research was framed as an exploratory case study of public RD&D financing of LCET in the Nordic countries. The study is constructed around indicator-based evaluation, confronting a literature review of indicators in RD&D policy evaluation with the actual usage of indicators in the Nordic countries. In order to enhance the understanding of indicator choice, an assessment of the indicator-based evaluation method was performed. Both primary and secondary data were collected to understand the respective funding instruments, their performance, and indicators used for their evaluation.[1]

10.2.1 Conceptualization of Indicator-Based RD&D Policy Evaluation

The multitude of indicators that is used in the assessment of RD&D policy can be categorized in different ways. A common differentiation is made between input, outcome, and impact indicators (Fischer 1995; Guedes et al. 2001; Neij and Åstrand 2006; Miedzinski et al. 2013). Another (complementary) approach to conceptualize the use of indicators is to view them as a way to operationalize criteria for policy evaluation (Mickwitz 2003). Relevant criteria that were used to structure this study are administrative capacity, effectiveness, and additionality.

It is debatable whether *administrative capacity* should be seen as an evaluation criterion as such or as a 'determinant of implementation' (Vedung 2000: 226). Following the IPCC (Intergovernmental Panel on Climate Change) (Kolstad et al. 2014), it was used as a criterion in this study. *Effectiveness* refers

[1] Further information about the case study, data collection and limitations of this study can be found in Sonnenschein (2016).

to the degree to which 'achieved outcomes correspond to the intended goals of the policy instrument' (Mickwitz 2003: 426). For this study also indicators were included, which were *de facto* used to assess the effects of a financing instrument but for which no explicit goals were formulated. Due to the large number of potential indicators for effectiveness this criterion is frequently subdivided into environmental effectiveness, technological progress, and commercial effectiveness (Carlsson et al. 2002: 243; Carley, Brown, and Lawrence 2012: figure 2). The *additionality* criterion complements effectiveness. It is the degree to which achieved outcomes differ from a baseline development that assumes the absence of the respective policy instrument. The challenge of attributing specific developments to individual policy instruments is large (Scriven 1991). Still, additionality is a core criterion to establish accountability for the success or failure of RD&D support policies.

10.2.2 Indicators Used in the Evaluation of LCET Support Policy

A comprehensive review of potential indicators used to assess LCET RD&D support is presented in Table 10.1. The table excludes social indicators and environmental indicators other than the ones related to greenhouse gas emissions. It differentiates between national level indicators and programme-level indicators and is structured according to the evaluation criteria presented in Section 10.2.1.

10.2.3 Key Indicators in the Context of Public RD&D Financing of LCET in the Nordic Countries

In order to reduce the scope of this study and increase its relevance, only the most salient indicators in the case study of LCET RD&D support in the Nordic countries were analysed. Moreover, only numeric indicators were chosen; and indicators included in the analysis had to be relevant at both national and programme level. Selected indicators included RD&D spending, CO_2 emissions, patents, commercial indicators (turnover, exports and jobs), return on investment (ROI), and the ratio of public and private RD&D. Further clarification about indicator choice follows (while the actual analysis of these indicators is presented in Section 10.3):

- The indicator 'CO_2 emissions', in this case, refers to CO_2 emissions from fossil fuel combustion determined with a production-based approach.
- The commercial indicators 'turnover', 'exports', and 'jobs' were grouped together as they are typically part of the same accounting system at the

Table 10.1. Indicators used in the literature on LCET support policies

	National level	Programme level
Administrative Capacity	*RD&D spending* *RD&D staff (and their formal qualification)*	*RD&D spending* *RD&D staff (and their formal qualification)*
	Effectiveness	
Environmental effectiveness	CO_2 emissions	CO_2 emissions
	CO_2 intensity of energy supply CO_2 intensity of the economy	
Technological progress	Patents (filed, granted, cited)	Patents (filed, granted, cited)
	Scientific papers (incl. PhD theses)	Scientific papers (incl. PhD theses)
	Learning rates	
	Technology/abatement costs	Technology/abatement costs
	Technology performance/ efficiency	Technology performance/efficiency
	Energy efficiency/intensity of the economy	
Commercial effectiveness	Jobs	Jobs
	Exports	Exports
	Turnover	Turnover
		Turnover/employee (productivity)
	Profits	Profits
	Return on investment	Return on investment
	Number of enterprises	
	Energy cost savings	Energy cost savings
Other	Energy self-sufficiency Share of renewable energy in energy supply	
	Additionality	
	Ratio of public and private RD&D spending	Ratio of public and private RD&D spending
	Jobs per energy output Net employment effect Macroeconomic multipliers	
		Scale and timing of private sector RD&D activity

Source: Author's compilation based on Stosic et al. (2016); Wilson et al. (2012); Carley et al. (2011) ; Carley et al. (2012); Gallagher et al. (2006); Neij and Åstrand (2006); Jacobsson and Rickne (2004); Spangenberg (2004); Kleinknecht, Van Montfort, and Brouwer (2002); Schoenecker and Swanson (2002); and Grupp (2000).

national level; also at the programme level they are often measured and presented together.

- As well as RD&D budgets, administrative capacity, in terms of knowledge and skills, was highlighted in the interviews as a key input factor for the success of public interventions, but did not seem to be reflected in evaluations. Input indicators related to administrative capacity, such as the number and qualification of fund managers and public officers in RD&D schemes (Gallagher, Holdren, and Sagar 2006), were not frequently used.
- Technological output indicators, other than patent counts, are bibliometric indicators and the number of supported PhDs, neither of which were explicitly used in this specific case and are potentially subject to large biases (Jacobsson and Rickne 2004).
- Only one additionality indicator was chosen to be part of this study, since no data on output and outcome additionality of publically financed RD&D programmes could be obtained.

10.2.4 Assessment of the Indicator-Based Method

Once indicators used to assess RD&D support to LCET in the Nordic countries were identified, categorized, and selected, they were analysed in order to assess the indicator-based evaluation method. The analysis focused on the *acceptance* of relevant stakeholders, on the *ease of monitoring* an indicator, including measurability and data availability, and on an indicator's *robustness* against manipulation. This evaluation approach was inspired by the 'RACER framework' for indicator choice in impact assessments (European Commission 2005). RACER stands for relevant, accepted, credible, easy to monitor, and robust. Both relevance and measurability were also suggested as criteria for the assessment and selection of green growth indicators (GGKP 2013).

The acceptance of an indicator was included in the analysis, since the results of an assessment that is based on poorly accepted indicators is not likely to resonate with key stakeholders and tends to have less policy impact. Moreover, indicators that are difficult to monitor or can only be monitored at very high costs are less likely to be applied in evaluations. The more expensive it gets to monitor the development of indicators, the harder it gets to justify resource use for evaluation. In contrast, the robustness of an indicator does not have immediate influence on programme evaluation, as less robust indicators can still be influential if they are widely accepted and monitored. Still, robustness is crucial from the academic perspective as indicators that are not robust may not provide conclusive indications for the (re-)design of LCET support schemes. Moreover, manipulation of indicators may eventually erode acceptance.

10.3 ANALYSIS: INDICATORS FOR PUBLIC RD&D FINANCING OF LCET IN THE NORDIC COUNTRIES

As outlined in Section 10.1, the development of the LCET sector in the Nordic countries is generally perceived as a success story. In contrast, the role of public RD&D financing in this story is more difficult to grasp as it has not been comprehensively researched. This study contributes to the evaluation of RD&D financing of LCET in the Nordic countries by scrutinizing the use of indicators rather than by presenting a comprehensive indicator-based evaluation as such. Hence, specific performance data from the case study is merely used to illustrate the use of indicators and their assessment.[2]

Indicators are analysed and ranked according to the criteria acceptance, ease of monitoring, and robustness (see Table 10.2). The estimation of indicators is presented on an ordinal three-point scale (zero, one, or two stars). The results represent the specific case of RD&D financing of LCET in the Nordic countries. Generalizability beyond LCET in the Nordic countries is particularly limited in the case of acceptance, while similar results can be expected for the criteria ease of monitoring and robustness if the study is repeated in a different context.

While most of the results are indicator-specific, there are some cross-cutting results, in particular with respect to robustness. First, the assessed performance may vary significantly depending on the definition of LCET, which is sometimes also referred to as green energy or clean energy technology. The decision to include controversial and capital-intensive technologies such as carbon capture and storage (CCS) or nuclear energy in the definition can make a large difference. Time-lags are another aspect that influences the robustness of indicators. While inputs into LCET RD&D are visible right away, outcomes and impacts of RD&D support programmes manifest themselves only after several years. Finally, for all aggregated indicators there is the challenge of attribution. It is virtually impossible to separate the effects induced by individual support schemes from other factors such as larger business cycles and general technological progress. In Sections 10.3.1–10.3.6, the schematic overview of results (Table 10.2) is substantiated for each of the six indicators.

10.3.1 RD&D Spending

10.3.1.1 Acceptance

RD&D spending is a widely accepted indicator in the Nordic countries. Policy makers have stressed the leading role of the Nordics in LCET RD&D by

[2] Further case-specific data and figures are included in Sonnenschein (2016).

Table 10.2. Overview of assessment indicators for public RD&D support to LCET, their acceptance, ease of monitoring, and robustness

Indicator	Acceptance	Ease of monitoring	Robustness
		Administrative Capacity	
RD&D budgets	**	**	*
		Effectiveness	
CO_2 emissions	*	*/**	*
Patents	*	**	**
Turnover, exports, jobs	**	*	o
ROI	o	*	o
		Additionality	
Ratio of public & private RD&D	**	*	*
		Legend	
	Indicator is:	*Indicator is:*	*Indicator:*
two stars (**)	*widely accepted by various stakeholders.*	*measurable and data is available.*	*is difficult to manipulate.*
one star (*)	*partially accepted by the stakeholders.*	*measurable but good data is not available.*	*can be manipulated but robustness can be tested.*
no star (o)	*only brought forward by one type of stakeholder.*	*not measurable.*	*is very prone to manipulation.*

Source: Author's analysis.

referring to budget allocations, academics have frequently used RD&D budget data in econometric studies of innovation activity, and public officers in LCET support programmes as well as fund managers stressed the particular role of public RD&D budgets for energy technology innovation in the interviews. At the level of individual programmes, larger public budgets are mostly, but not always, perceived as desirable. The success of commercialization support programmes, for instance, largely depended on the existence of suitable innovative enterprises. By increasing budgets and, hence, the number of supported enterprises, the risk of picking less-promising enterprises increases.

10.3.1.2 Ease of Monitoring

Comprehensive data on national energy RD&D spending of Nordic countries is reported to and published by the International Energy Agency on an annual basis (IEA 2015b). The resolution of the data is fine enough to differentiate between LCET and other energy technologies. RD&D spending data is also available at the programme level, even though it is scattered, so that it requires some data-gathering effort to obtain a systematic overview.

10.3.1.3 Robustness

The particular presentation of RD&D spending data leaves room for manipulation. Today's share of LCET RD&D in GDP is, for instance, very high in the Nordic countries compared to other industrial states (IEA 2015b). However, when comparing to historic data, it was three times higher in Sweden in the early 1980s, which has to be seen in the context of the oil crises (IEA 2015a).

RD&D spending on LCET can also be compared to overall public RD&D spending, which represents about 3–3.5 per cent of GDP in the Nordic countries (as compared to 0.03–0.11 per cent for LCET). At the time of the oil crises, energy R&D made up more than 10 per cent of overall R&D both in Europe and the Americas, a ratio that has dropped to 2 per cent and 3 per cent respectively (IEA 2015a).

Absolute RD&D spending is very low in the Nordic countries as compared to larger countries. The US loan guarantees of US$535m to solar cell producer Solyndra and of US$465m to electric car manufacturer Tesla (Rodrik 2014) exceeded the current capacity of the Nordic countries' RD&D budgets, which seem even smaller in comparison to the support that China grants to some of its renewable energy companies, for example US$9.1bn to LDK Solar, US$7.6bn to Suntech Power, and US$7bn to Yingli Solar (Sanderson and Forsythe 2013).

Moreover, RD&D financing for LCET may well be concentrated in a few lighthouse projects, as, for example, CCS funding is in Norway, which made up more than half of RD&D to LCET between 2009 and 2012 (IEA 2015b). In contrast, it was observed that there was too little public funding for early stage enterprises that have already received seed-funding but often have difficulties securing follow-up financing (Grünfeld, Iverson, and Grimsby 2011).

Finally, RD&D spending is not adjusted for the respective costs of conducting RD&D, for example the costs for employing research staff, which are significantly higher in countries like Sweden as compared to many other European countries (Jacobsson and Rickne 2004).

10.3.2 CO_2 Emissions

The most apparent indicator for assessing the environmental effectiveness of public RD&D financing of LCET in the Nordic countries is the development of CO_2 emissions from fossil fuel combustion. It is often presented in relation to GDP growth in order to account for the size of the respective economy.

Both CO_2 emissions and the emissions intensity of the economy fell in all Nordic countries but Norway between 2000 and 2014 (IEA 2015c).

10.3.2.1 Acceptance

CO_2 emissions are not widely accepted as a significant impact indicator. On the one hand, investigated policy programmes and the laws in which they are enshrined do refer to the reduction of CO_2 emissions, and also academics comprehensively discuss the role of technology push policies for reducing CO_2 emissions. On the other hand, emission reductions do not play a major role at the programme and project level. The interviews revealed that the reduction of CO_2 emissions is seen as a 'by-product' of the (economic) success of supported enterprises and not as an indicator for success in itself.

10.3.2.2 Ease of Monitoring

CO_2 emissions data is certainly measurable and available at the national level in the Nordic countries but difficult to measure at the programme level, as the lion's share of emission reduction typically does not take place in RD&D projects but indirectly through selling and deploying LCETs on domestic and international markets. Only few programmes included CO_2 emissions in their assessment, for example Enova Norway's support for 'new energy technology', which monitored energy savings and CO_2 emission reductions both in absolute terms and in relation to provided funding (Enova 2015).

10.3.2.3 Robustness

While national-level emissions data is rather robust and an established system for monitoring, reporting, and verification is in place in all Nordic countries, there is a lot of room for manoeuvre at the programme level. Either direct or induced emissions reductions may be monitored, at both the national and international level. Moreover, the choice of the baseline for evaluating reductions, and not merely monitoring them, leaves room for manipulation. Base years may vary and business as usual scenarios rest on many assumptions.

10.3.3 Patents

All Nordic countries multiplied their share of low-carbon technology patents in total patents between 1999 and 2011, reaching about 10 per cent in 2011 (OECD 2015). This suggests that within the Nordic countries LCET became a more significant area of innovation, which may be partly driven by additional public RD&D financing in this sector. This trend is not restricted to the

Nordic countries, but it is likely more pronounced than in many other countries, so that the 'relative technological advantage' of Nordic countries in LCET may well have strengthened in this period (Haščič and Migotto 2015: 30).

10.3.3.1 Acceptance

At the national level, patents are frequently used as proxies for technological progress, both by academics and government agencies. The situation is different at the programme level where patents are mainly regarded as a means to an end. Even if not seen as ends in themselves, patents and the process of protecting intellectual property rights do play a role in the RD&D support that is provided to LCET in the Nordic countries. Patents are simply not regarded as a relevant indicator for success at the programme level.

10.3.3.2 Ease of Monitoring

Patent data of LCET is available at the national level and published regularly. In contrast, patent data is not made available in a systematic way at the programme level, so that the attribution of patents to public support instruments becomes difficult. The Finnish national innovation funding agency TEKES monitors the overall number of patents registered by supported organizations but does not provide a specific breakdown for LCET (TEKES 2015). A Danish study of the green economy compares innovation activity and patenting of green enterprises to all enterprises, showing that the trading of patents and intellectual property rights plays a larger role in green enterprises than in the overall economy (Danish Energy Agency 2012: 38).

10.3.3.3 Robustness

Patents are a robust indicator. Data is available, it can be rather easily verified so that there is little room for manipulation, and patents can to some extent be attributed to RD&D projects. Still there is a risk that funding agencies account for a full patent in cases in which they provided only a minor share of the overall project budget.

10.3.4 Turnover, Exports, Jobs

Turnover in the LCET sector, its jobs and exports are frequently used indicators in the context of RD&D financing instruments. The most developed and standardized way to measure the commercial development of subsectors of the green economy is provided in national statistics about the Environmental

Goods and Services Sector (EGSS), which is defined in the statistical guidelines of Eurostat (Eurostat 2009).

10.3.4.1 *Acceptance*

Various stakeholders stress the commercial dimension of RD&D financing of LCET. There is virtually no public support programme in the Nordic countries that does not explicitly refer to economic development. The political emphasis of commercial aspects is a view that was reaffirmed in the interviews where public officers stressed the role of commercialization potential. Even in academia the focus is increasingly put on the commercialization aspect of publically funded RD&D (Jacobsson, Lindholm-Dahlstrand, and Elg 2013).

10.3.4.2 *Ease of Monitoring*

Turnover, jobs, and exports in the LCET sector are measurable and some data is available at both the national level (in statistics on the EGSS) and at programme level. However, available data is scattered and cross-country comparisons are not possible. Sweden is the only Nordic country that has collected comprehensive data on its EGSS for more than a decade, including specific data on the subsectors renewable energy, and energy savings. The Danish EGSS statistics only cover the years 2012–14, the Finnish statistics do not include the subsectors renewable energy and energy efficiency, yet, and in Norway the statistics office is preparing for the first publication of EGSS data in 2017. The lack of official data from statistics offices is partly compensated for with data from industry associations (Mellbye and Espelien 2013; Cleantech Finland 2014).

Specific programme evaluations sometimes also include the economic outcomes of RD&D support programmes. The Danish Business Innovation Fund, which financed mainly green economy enterprises in 2010–12, required, for instance, all supported enterprises to communicate five-year turnover and employment targets. These targets were summarized and followed up in a mid-term evaluation (Deloitte 2012), but no further evaluation with actual data is available, yet. This example illustrates a typical challenge of programme evaluations. Once temporary support programmes are finalized, little priority and resources are given to evaluation.

10.3.4.3 *Robustness*

As well as data availability, quality of commercial data also varies. Due to the fact that there is no standardized way to measure commercial indicators for LCET (and the whole EGSS) the data may vary between different sources.

In particular, data from grey literature tends to be less robust. One example is Norway's renewable energy sector, for which industry sources frequently report employment of 50,000, a turnover of NOK200 bn and approximately 2000 companies in 2010. This is far higher than the figures published in a more elaborated study, which found 13,700 employees, NOK85bn turnover and 860 companies in 2010 (Mellbye and Espelien 2013).

Furthermore, economic data about the LCET sector does not reflect that employment, turnover, and exports could also be generated in other sectors. The actual figures do not reflect the net effect of the respective support policies, that is its additionality, but only their gross effects. The claim that the Danish wind power sector employs more people than the Swedish automotive industry is often made in the context of job creation. This is potentially misleading as it does not say anything about the net employment effects of past wind power support policies in Denmark.

10.3.5 Return on Investment (ROI)

In the case of public equity financing instruments, ROI is an additional commercial indicator under consideration. There is no exclusive public venture capital (VC) fund for LCET in the Nordic countries, but several public VC funds have LCET companies in their portfolio. These funds typically stress that they operate like private funds and that their main objective is ROI. This supports the findings of Yang and Sollen (2013), who found strong evidence for a *de facto* profit motive in state-owned VC in the Nordics.

The track record of public VC to LCET enterprises has a rather poor image among analysts in the Nordic countries, some of whom call it a complete absence of success stories. Due to 'poor financial returns on Cleantech investments' (Murray and Cowling 2014) the Danish Growth Fund has not made any initial VC investments into cleantech since 2011, and neither has the Norwegian public VC fund Investinor. Even the performance of private VC funds that invest in cleantech is at best mixed in the Nordic countries (Wang 2015). The absence of success may have other reasons than public VC being an inappropriate support instrument, including the poor timing of investments with respect to economic cycles and long lead times in this sector, which means that there have not been many exits, yet (Murray and Cowling 2014).

10.3.5.1 Acceptance

With the exception of (state-owned) VC fund managers, little support could be gathered for taking ROI into consideration as an indicator for the effectiveness of public RD&D financing of LCET. Several stakeholders argued that the state should support those ventures that are too risky for the private sector but potentially beneficial for society. These are most likely not the ones that promise the highest returns. While it is widely accepted that profit-orientation should be the *modus operandi* for public equity funds, a general profit target is not accepted at all. In the interviews it was suggested that benefits to the state could be assessed in a different way, that is by looking at financing costs and at the indirect impact on tax revenue that is triggered by additional commercial activity.

10.3.5.2 Ease of Monitoring

The returns from public VC investments into LCET are measurable, which is straightforward after a portfolio company has been sold (exit). There are, however, large methodological challenges in estimating the current value of existing portfolios. LCETs have a long time to market so that several of the public investments in the Nordic countries could not be exited yet, which impedes the calculation of ROI. Good data for public VC investments in the Nordic countries is not available, and even less so for LCET investments in particular, since LCET investments are typically part of larger VC funds that are not specialized into energy or cleantech.

10.3.5.3 Robustness

Due to the lack of data, it is not possible to assess the actual robustness of the indicator ROI. Still, it is rather clear how the data could be manipulated and why. Fund managers have strong incentives to overestimate the current value of their portfolio, while entrepreneurs also have to portray their respective ventures as a success story in order to receive continued financing.

10.3.6 The Ratio of Public and Private RD&D Financing

Merely looking at effectiveness is not sufficient to assess the success of a policy intervention. RD&D financing instruments in the Nordic countries showed a clear attempt not only to be effective but also to both ensure the additionality of the intervention and, to a lesser extent, monitor this additionality effect. The most common indicator for the additionality of Nordic RD&D support schemes was the ratio of public and private RD&D financing, that is the consideration of whether public financing has crowded in or crowded out private financing.

10.3.6.1 Acceptance

The ratio of public and private RD&D was clearly the indicator that was used most to investigate additionality. Its role as input indicator, however, slightly reduced acceptance, as after all additional effects on the environment, technology, and economy were sought after rather than additional financial input, as such. Furthermore, the indicator is not always easy to interpret, which further reduced acceptance.

10.3.6.2 Ease of Monitoring

Both public and private RD&D financing are measurable and data is partially available, even though data on private sector RD&D spending is less comprehensive. There is no comprehensive study about private versus public sector RD&D for LCET in the Nordic countries. At the programme level the evidence from evaluations, reports, and interviews clearly suggests very high additionality of public RD&D to LCET in the Nordics. Gaps in the innovation financing cycle of cleantech were identified by various private and public investors (Finnsson 2011). Evaluations of TEKES' (Finland) financing of environmental technology (Valovirta et al. 2014) and of Innovation Norway's Environmental Technology Scheme (Espelien et al. 2014) found high degrees of additionality. In the latter case NOK1 of financing 'triggered' NOK3.6 in private investments. Moreover, in the case of Sweden, public funding seems to crowd in private capital for cleantech investments; and co-investments are particularly common in the sub-sector of energy (Yang and Sollen 2013: 59). While, at the programme level, data on private co-investments in RD&D is collected and, in many cases, even has to be collected, this does not provide any information about private RD&D activity outside publicly co-financed projects.

10.3.6.3 Robustness

The ratio of public and private sector RD&D financing is a simple input indicator and as such it avoids some of the difficulties in assessing the additionality of programme outcome. However, interviewees pointed out that the interpretation of the indicator is not self-evident. Additionality of public funds is likely if the share in total RD&D financing (public and private) remains the same or even decreases. It is more difficult to interpret when the share of public RD&D increases. This could be either due to a crisis in private RD&D financing, hence pointing towards a high degree of additionality, or due to crowding out, indicating a low degree of additionality.

One example for an increased share of public financing is the development of cleantech VC in Sweden. Private VC cleantech investments in Sweden dropped from their peak at nearly 700 million SEK in 2008 to about 50 million SEK in 2014, while dedicated public VC funds increased from about 25 million SEK to 100 million SEK (Tillväxtanalys 2015). The collapse of private VC investments despite slightly increasing dedicated public VC funds provides indication for a high degree of additionality of public funds, even though they were not successful in crowding in much private funding.

Besides its ambiguity, the robustness of this indicator is further challenged by the fact that private sector RD&D financing data is largely based on self-reporting. Companies have many options for manipulating the data they report, for example increasing their budgets by inflating the staff hours they put into an RD&D project.

10.4 EFFECTS OF INDICATOR CHOICE: POTENTIAL BIASES AND THEIR POLICY IMPLICATIONS

The analysis of six common indicators in the context of RD&D financing of LCET showed that even a basic structured assessment does not result in a clear-cut indicator-based evaluation framework. Trade-offs between comprehensiveness, acceptance, ease of monitoring, and robustness are impossible to avoid.

An argument in favour of indicator-based monitoring and evaluation is that it helps to establish accountability of policy makers. If evaluations of RD&D programmes for LCET are carried out at all, they are typically based on indicators. Accepting that these indicators only represent a subset of all available indicators, moreover a subset that is faced with heavy trade-offs, it becomes clear that the mere selection of indicators can have a major impact on evaluation results. These results then feed back into the policy-making process and may trigger changes in programme design and strategic focus.

The active selection of indicators may introduce bias into indicator-based evaluation. It is important to note, though, that certain biases might be justified as the specific objectives of different programmes (e.g., technological progress or economic growth) may differ. Flexibility in the computation and presentation of indicators (i.e., lack of robustness) introduces further uncertainty about the validity of assessment results. In Section 10.4.1 to 10.4.3, some potential biases and uncertainties in the evaluation of Nordic RD&D financing of LCET are discussed and possible policy implications are mentioned.

10.4.1 A Focus on Short-Term Economic Performance and ROI

In the case study, the growth of jobs, exports, and turnover, and also profitability appeared to be increasingly important indicators of the public financing of RD&D and its commercialization. At the same time private RD&D spending in the Nordic LCET sector has recently decreased and public RD&D

spending has levelled off. The interviews made clear that, in particular, public VC instruments have moved away from cleantech due to low profit expectations and long times to market. Hence, dedicated support for LCET is not likely to perform well in assessments if much attention is paid to the indicators ROI and the (short-term) development of jobs, exports, and turnover.

A bias towards these indicators largely disregards social benefits related to the development and deployment of LCET, such as resource conservation and climate change mitigation. This improves the position of other sectors in the competition for public funds. The information and communication technology (ICT) sector is, for instance, less capital-intensive and has shorter development cycles.

However, LCET-specific support and commercial success do not exclude each other in the Nordic countries. Analyses of the Danish wind energy sector and the Swedish bioenergy sector have shown that 'medium-sized countries can be within the world's leading nations in a specific field of energy technology, if appropriate supply and demand side policies support a certain technology' (Bointner 2014: 738). In order to be commercially successful, public RD&D financing of LCET likely has to be part of a more comprehensive policy mix. Accordingly, fund managers and public officers stressed in the interviews that the business plans of several supported companies could only be worked out if demand side policies were in place. Demand-side measures include feed-in-tariffs for renewable energy in Denmark and Finland, the common green certificates market of Norway and Sweden, CO_2 taxes, and deployment subsidies for various LCETs. The main policy implication of a strong focus on short-term economic performance is, hence, that (further) dedicated support to LCET is difficult to justify if there are no additional demand side policies in place.

10.4.2 Stressing the Additionality of Financing

Despite the lack of workable indicators, additionality was strongly emphasized in both interviews and reports. This was slightly surprising as there was virtually no evidence for 'crowding-out' private capital from the Nordic LCET sector. The perceived importance of additionality can be traced back to regulatory requirements stipulated in EU state aid regulation. The investigated support instruments included various institutional mechanisms to make sure that the state does not finance 'too much', including co-investment provisions, maximum aid intensities, and limited opportunities for follow-up investments.

It would be an exaggeration, though, to understand these mechanisms as a result of a bias towards additionality in evaluation. While the importance of additionality was indeed frequently stressed, actual monitoring happened, if at all, mainly for the input indicator 'ratio of public and private funding'. This supports the thesis that 'additionality can be treated ex ante as a design criterion and ex post as an area where some evidence can be collected but where full measurement may be impossible and in any case is not justified in resource terms' (Georghiou 2002: 64). It would require further discourse analysis to better understand how the frequent discussion of additionality has influenced the design of RD&D financing measures in the Nordic countries.

The main policy implication of a large emphasis on additionality indicators in the assessment of RD&D support schemes is that it may favour cautious state intervention rather than strong industrial policy push for LCET.

10.4.3 Disregarding Decarbonization

Decarbonization was a very prominent objective in the justification and communication of LCET support measures in the Nordic countries, while at the programme level climate aspects were overtrumped by innovation objectives. Accordingly, most RD&D support to LCET companies was managed by dedicated innovation agencies like TEKES, Innovation Norway, and Vinnova.

The challenge to place LCET support within a certain policy domain is well-illustrated by an evaluation of the Norwegian Environmental Technology Scheme (Innovation Norway). The hierarchy between the scheme's objectives 'environmental effect' and 'commercial potential' was not clear and the evaluators recommended 'design[ing] explicit objectives including a clear goal hierarchy as soon as possible' (Espelien et al. 2014: 6), being very outspoken that priority should be given to commercial potential. This reflects a frequently expressed view in the case study, that is that commercial success is the best strategy to assure positive environmental impact.

Moreover, previous econometric studies have shown that little direct influence of public RD&D financing on CO_2 emissions from energy can be expected (Garrone and Grilli 2010). Considering further that there are serious methodological challenges to attribute emission reductions to specific RD&D support schemes (Miedzinski et al. 2013), it was not surprising that the indicator 'CO_2 emissions' was largely disregarded in evaluations of the analysed instruments.

The potential policy implication of disregarding CO_2 emissions as an assessment indicator lies in the selection of LCETs that are worth supporting. There is a risk that the mitigation potential of a technology becomes secondary concern in the selection of support-worthy RD&D projects and enterprises.

10.5 CONCLUDING REMARKS

The main objective of this study was to assess the performance of indicator-based evaluation in the context of public RD&D financing of LCET. The

Nordic countries provided an interesting case to study the choice of indicators in policy evaluation, their acceptance, the ease of monitoring them, and their robustness. The analysis clearly showed that a structured assessment of indicators can help to point up the trade-offs and limitations that are inherent in indicator-based evaluation. Selecting indicators can introduce bias. The discussion of LCET RD&D financing in the Nordic countries illustrated how a focus on short-term economic performance may hinder (further) dedicated support to LCET, how stressing the additionality aspect of public financing may lead to rather cautious state intervention, and how the partial neglect of CO_2 emissions in evaluation may shift the focus away from the abatement potential of supported technologies.

If such biases happen to correspond with the policy objectives behind the respective instruments and programmes, they can be justified. If, on the other hand, the ambition is to act according to the targets of the 2015 Paris Agreement, more dedicated support to LCET with substantial abatement potential and bold state interventions are needed.

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Paper III

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ORIGINAL ARTICLE

Minimum energy performance standards for the 1.5 °C target: an effective complement to carbon pricing

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Abstract Radical energy efficiency improvements are needed to keep global warming within 1.5 °C until the end of the century. Minimum energy performance standards (MEPS) are a widely applied policy instrument to improve the energy efficiency of appliances and reduce CO_2 emissions, but they are criticized as redundant if an overarching carbon pricing scheme is in place. In order to better understand how MEPS could play a more effective role in reaching the 1.5 °C target, life cycle costs (LCC) for four home appliances were modelled considering a cost for emitting CO_2. First, a significant social cost of carbon was introduced in a LCC optimisation model and it was found that a modest tightening of MEPS is sufficient to account for the climate externality. Second, more stringent MEPS were modelled and it was found that the switching prices needed to incentivize a shift up one or two efficiency classes were far higher than current carbon prices. These results have several implications for climate policy towards the 1.5 °C target. MEPS can easily internalize the climate externality and have the advantage over carbon pricing that policy makers can be certain that consumers actually move to more efficient appliances. While stringent MEPS do not appear to be economically efficient on the short-run, they are likely cost-effective in long-run 1.5 °C-consistent scenarios.

Keywords MEPS · Carbon pricing · Social cost of carbon · Life cycle costs · Appliances · 1.5 °C target

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Introduction

Energy efficiency improvements are crucial for limiting global warming to 1.5 °C by 2100, which is the aspirational target of the Paris Agreement (United Nations, 2015). A review of 1.5 °C-consistent scenarios found that 'returning warming to below 1.5°C by 2100 becomes infeasible if final energy demand is not kept to very low levels' (Rogelj et al., 2015a, p. 526). Despite economic and population growth, global final energy consumption has to be roughly kept at the current level of about 8 Gtoe/year to maintain chances to stabilize global warming at 1.5 °C by the end of the century (Akimoto et al. 2017). In contrast, scenarios with significantly increasing energy demand are not compatible with reaching the 1.5 °C target (Rogelj et al., 2015b). Moreover, high-energy demand scenarios are associated with high total mitigation costs (Bertram et al., 2015a; Rogelj et al., 2016).

A large and rapidly growing area of energy consumption is home appliances (Cabeza et al. 2014). Market forecasts indicate annual growth rates of the market for home appliances of 6% until 2022 (Oristep Consulting, 2017) and growth in the sales of white good units from

640 million in 2016 to 850 million in 2021 (Kithany et al., 2017). If materialized, such growth is not compatible with 1.5 °C-consistent scenarios unless significant improvements in energy efficiency are realized. Current appliances vary largely with respect to their energy efficiency, with the best available technology (BAT) often being 30–50% more efficient than what regulatory standards require (Lucon et al., 2014). By cutting off the worst performing appliances and targeting BATs, there is potential to reduce global annual CO_2 emissions by 13% in 2030 (Letschert et al., 2013, p. 80), thereby making a significant contribution to achieving the 1.5 °C target.

Minimum energy performance standards (MEPS) are a central policy instrument to promote energy efficiency in home appliances by banning the worst-performing appliances from the market, thereby forcing manufacturers into innovation and consumers into the adoption of more energy-efficient technology (Sachs, 2012; Siderius, 2014). For the EU, MEPS set under the overarching Ecodesign Directive are estimated to deliver 991 TWh of energy savings in the residential sector alone by 2020 (VHK 2016). In 2010, the products covered by the Directive were responsible for 1955 Mt CO_2e of GHG emissions, 41% of the total EU-28 emissions (VHK 2016). In 2030, the emission reduction for the average product is estimated to be 30% vs. business as usual, implying a reduction of approximately 11% of the EU total GHG emissions (VHK 2016). This abatement potential is remarkable when considering that MEPS are not primarily climate policy instruments. While the EU's standard setting process includes life cycle analysis (LCA), and thus considers climate aspects, the analytical determination of MEPS is generally made by determining which efficiency requirement leads to an overall minimum life cycle cost (LCC) for end users (EU 2009, Article 15). LCC includes purchase price and running costs, but climate externalities are generally not included.[1]

In order to account for the role that MEPS could play in reaching the 1.5 °C target, one approach is to consider these climate externalities—the so-called social cost of carbon (SCC)—in LCC modelling of home appliances. Similarly, the shadow price of carbon associated with 1.5 °C-consistent climate mitigation scenarios could also be considered in LCC modelling. From a methodological perspective, the integration of shadow prices in LCC modelling is equivalent to the integration of SCC. In this study, mainly the term SCC is used, but differences to shadow carbon prices are noted whenever relevant. Accordingly, the first objective of this study was to show how SCC can be integrated into LCC modelling of home appliances and to present the potential implications for setting MEPS. For this purpose, the LCC optima with and without SCC were identified for four different home appliances (refrigerators, dishwashers, tumble dryers and televisions) across different energy efficiency classes.

The role of MEPS in the climate policy mix is controversial. Despite the existing track-record of MEPS around the world (Molenbroek et al. 2015), putting a price on carbon through taxes and emission trading schemes (ETS) has been argued as the first-choice policy to deliver cost-effective abatement and innovation incentives by internalising climate externalities (Aldy and Stavins 2012; Goulder and Parry 2008). The overlap of MEPS and emissions trading schemes has been argued to be economically inefficient, as emissions are not abated where the market finds it cheapest but where mandated by policy makers (Böhringer et al. 2016). That said, there is evidence suggesting that efficient home appliances are among the cheapest abatement options (Hood 2013; Wada et al. 2012), but this abatement potential has been underutilized so far, indicating market failures and behavioural anomalies (Gerarden et al. 2017; Gillingham and Palmer 2014). MEPS, in turn, have been argued to address several of these failures (Houde and Spurlock 2016; Schleich et al. 2016), which may even increase the overall economic efficiency of emissions abatement by ensuring that the low-cost abatement potential in this area is utilized (Hood 2013).

The cost-effectiveness of policies targeted at home appliances can be assessed by looking at their marginal abatement costs in comparison to other low-carbon energy technologies. In the context of technology pathways for ambitious climate change mitigation, such as the 1.5 °C target, specific 'switching prices' have been calculated, i.e. the carbon prices required to incentivize a shift towards low-carbon energy supply technologies, such as renewable energy or carbon capture and storage technology (IEA 2016a; Stiglitz et al. 2017). While on the demand side there seems to be general evidence for low abatement costs of energy efficiency technology, in the case of home appliances, evidence for *specific* switching prices is lacking. Thus, the second objective of this study

[1] Indirectly, climate aspects might be partly considered via existing carbon pricing instruments.

was to investigate the switching prices that correspond in their incentive effect to stringent MEPS. For this purpose, carbon prices were estimated that would be needed to make the LCC of less efficient, but cheaper, appliances on the market at least as high as the LCC of more efficient and more expensive appliances. This would then be the minimum requirement for consumers who fully consider LCC in their purchase decision to switch to a higher efficiency class.

The empirical case that was used to address the two research objectives was the UK market for home appliances in 2016. While the analysis was focused on the UK, the sensitivity of results was tested with respect to key factors that may differ in other countries, including the CO_2 emissions factor of the electricity mix and the electricity price, making the results applicable in different contexts. The results of this empirical study have implications for designing ambitious MEPS as part of a deep decarbonisation policy mix consistent with 1.5 °C warming by 2100.

The article is structured as follows. First, previous research on SCC is introduced and the method that was used to model the LCC of appliances is described. Second, quantitative results of LCC modelling with and without SCC are presented. Special attention is given to the robustness of these results regarding changes in key assumptions. Third, the methodological novelty and limitations are discussed. Finally, implications for 1.5 °C-consistent energy efficiency policy are highlighted.

Research design and methods[2]

Estimations of the social cost of carbon and shadow carbon prices in 1.5 °C scenarios

In theory, an efficient carbon price takes into consideration estimates of damages from climate change in the form of an SCC and marginal emission abatement costs (MAC). As long as the MAC does not exceed the SCC, further abatement efforts should be undertaken, as they are beneficial from a societal perspective (Aldy and Stavins 2012). However, estimating the costs and benefits of climate change mitigation involves many uncertainties and assumptions (Arent et al. 2014; Nordhaus 2007; Schelling 1992; Stern 2007). With respect to electric home appliances, carbon prices increase LCC by pricing the carbon content of fuels used in generating the electricity that is used by the appliances. In turn, the SCC reflects the social costs of CO_2 emissions associated with the use of electric appliances.

Estimates for the SCC vary from one digit values (in USD) per ton CO_2 (Tol 2005) to several hundred (Moore and Diaz 2015) and even over a thousand (Ackerman and Stanton 2012). The central US Government SCC estimate is 43 USD/tCO_2 in 2020, assuming a social discount rate of 3% (Revesz et al. 2014). Prior to 2009, the UK Government used an SCC estimate of USD 83, based on the Stern Report (Stern 2007). At the top of the spectrum, Sweden bases several of its policies on a SCC estimate of more than USD 130 (Trafikverket 2016). While much research needs to be done on the SCC (Burke et al. 2016), a range up to 150 USD/tCO_2 covers most of the current estimates.

Recent energy-economy modelling suggests, on the other hand, that the global shadow price for carbon in 2030 centres around 100 USD/tCO_2e in scenarios consistent with the 2 °C target (Clarke et al. 2014; Guivarch and Rogelj 2017) and 200–300 USD/tCO_2e in 1.5 °C scenarios (Rogelj et al., 2015a). While SCC reflects costs associated with climate change, these shadow prices indicate mitigation costs. It is important to note that shadow prices can represent various actual policy instruments, including carbon taxes and ETS, but among others also technological regulations, mandates or subsidies, which are all associated with different implicit carbon prices (Guivarch and Rogelj, 2017).

UK appliance market data

The appliance data that were used in LCC modelling (incl. appliance price, electricity use, product features, efficiency rating) stem from online marketplaces for energy-efficient appliances[3] and reflect the market offering in the UK in 2016. Table 1 provides an overview of this data, comparing it with actual sales data for products in the respective categories in the EU. Note that the sales breakdown for the UK may well be different than for the EU and that sales might have shifted between 2013 and 2016.

[2] A spreadsheet with data and analysis is available in the supplementary material of the online version of this article.

[3] Enervee market, https://enervee.com/ and http://www.johnlewis.com/ in the UK.

Table 1 Description of the dataset by appliance type and efficiency class, including sales shares in EU for latest available years

Appliance (current MEPS)	n =		A+++	A++	A+	A	B	C
Refrigerators (A+)	978	Number of models	37 (4%)	317 (32%)	624 (64%)	0 (0%)		
		% sales in EU 2015	5%	25%	68%	2%		
Dishwashers (A+)[a]	358	Number of models	54 (15%)	89 (25%)	184 (51%)	31 (9%)		
		% sales in EU 2013	3%	23%	35%	38%		
Tumble dryers (B)[b]	148	Number of models	4 (3%)	49 (33%)	13 (9%)	0 (0%)	63 (43%)	19 (13%)
		% sales in EU 2015	4%	28%	14%	1%	33%	19%
Televisions (D)[c]	189	Number of models	0 (0%)	2 (1%)	79 (42%)	89 (47%)	19 (13%)	0 (0%)
		% sales in EU 2013	0%	1%	23%	45%	13%	3%

[a] In Fig. 2 we graph only models with 12 place settings as 82% of models sold in the EU in 2009 were 12 place settings (European Commission 2010). The category with a capacity of 12 place settings includes 57 models in the Enervee dataset

[b] Class C includes slim models. In Fig. 2 we graph only models with 8kg capacity (the median size) with 60 models in that category

[c] 16% of sales were of unknown energy class

Source: Michel et al. (2013, 2016); VHK et al. (2014)

Adjusting product prices and energy efficiency to product characteristics

Before starting the actual LCC modelling, the market data regarding appliance price and annual unit energy consumption (UEC) had to be prepared for analysis in order to separate the effect of efficiency from the effects appliance size and other product features have on price and UEC. For that purpose, a regression analysis was performed independently for price and UEC (Van Buskirk et al., 2014—supplement). Fixing product attributes to their average market value and then evaluating the regression function results for both price and UEC as a function of efficiency level provides the price vs. UEC relationship, factoring out the influence of changes in attributes between efficiency levels. The key assumption of the method is that the function that describes price and UEC as a function of attributes and efficiency level can be determined using regression analysis from market data.

For all four appliance types (dishwasher, refrigerator, tumble dryer and television), controlled variables included size (capacity or screen size) and efficiency class. In order to test the robustness of this method, a more comprehensive regression was performed for televisions, an appliance type where difficulties of using LCC approaches have occurred previously due to a lack of clear relation between price and energy efficiency (Siderius, 2013). The additional product features that were controlled for in the case of televisions were NFC (near-field communication), smart television, screen type, screen resolution and number of tuners.

LCC modelling

Once the relationship between price and UEC was established, LCC could be computed. The LCC of an appliance is the sum of its price and the present value of operating costs. In modelling LCC and estimating the LCC optimum for four types of appliances, the approach of LCC optimisation outlined in Van Buskirk et al. (2014) was used.[4] LCC is defined as follows:

$$\mathrm{LCC} = P_A + \mathrm{PWF} \times P_E \times \mathrm{UEC} \tag{1}$$

where P_A is the total average appliance price for one efficiency class, and UEC is the average annual unit energy use in the respective class. P_A and UEC are corrected for product characteristics as described above. P_E is the price of electricity, and PWF is the present worth factor:

$$\mathrm{PWF} = \frac{1-(1+i)^{-L}}{i} \tag{2}$$

where i is the discount rate and L average product lifetime. When including the SCC, the price of

[4] The LCC optimization method is only briefly presented here; for a full explanation, please refer to the "Supporting Information" in Van Buskirk et al., 2014

electricity is increased by the product of the SCC and the emission factor (EF).[5] The revised Eq 1 for LCC including SCC is as follows:

$$\mathrm{LCC}_{\mathrm{SCC}} = P_A + \mathrm{PWF} \times (P_E + \mathrm{EF} \times \mathrm{SCC}) \times \mathrm{UEC} \quad (3)$$

Estimating the LCC optimum with and without SCC

Once the relation between appliance price and UEC has been established, and if there is a clear trend that lower UEC implies higher appliance prices, LCC optima can be computed. As the purchase price of an appliance increases without bound when energy use decreases towards the theoretical minimum, and energy operating costs increase without bound for very low efficiencies, the minimum LCC of an appliance can be usually found somewhere in between the most efficient and the least efficient appliances. Additionally, when appliance prices increase with decreasing UEC, then the LCC vs. UEC relationship is typically as illustrated in Fig. 1 (Van Buskirk et al., 2014). The minimum in the LCC function theoretically determines the optimum value for MEPS.

Focussing on the LCC function near the minimum value, the LCC vs. UEC function can be approximated as a quadratic function[6]:

$$\mathrm{LCC}(\mathrm{UEC}) = \mathrm{LCC}_{\min} + C \times (\mathrm{UEC}-\mathrm{UEC}_{\min})^2 \quad (4)$$

where $\mathrm{LCC}_{\min}$ is the LCC value at the minima, and $\mathrm{UEC}_{\min}$ is the energy use corresponding to the minimum LCC value, and C is a constant that describes the curvature of the LCC vs. UEC curve near the LCC minimum. When internalising the climate externality with the SCC, the price of electricity increases and the UEC for the minimum LCC shifts to a lower value. Similar to Eq. 4, LCC can now be modelled as a function of SCC and UEC in the following way:

$$\mathrm{LCC}_{\mathrm{SCC}}(\mathrm{UEC}) = \mathrm{LCC}_{\min} + C \times (\mathrm{UEC}-\mathrm{UEC}_{\min})^2 + \mathrm{PWF} \times \mathrm{EF} \times \mathrm{SCC} \times \mathrm{UEC} \quad (5)$$

The minimum LCC can now be calculated by taking the derivative of the right-hand side of Eq. 5 with respect to UEC, setting it equal to zero and solving for the minimum UEC as a function of SCC and other parameters. This results in the following equation that describes the shift in UEC if SCC is included in LCC optimisation:

$$\mathrm{UEC}_{\mathrm{SCC\_min}} = \mathrm{UEC}_{\min} - \frac{\mathrm{PWF} \times (\mathrm{EF} \times \mathrm{SCC})}{2 \times C} \quad (6)$$

This equation describes that near the old LCC minima, the shift in optimum UEC due to consideration of SCC is proportional to the value of SCC, the EF, and the PWF, and the shift is inversely proportional to the curvature of the LCC vs. UEC curve (i.e. C) near the LCC minimum. Note that this modelling approach can be used when the available data shows an LCC minimum that exists within the range of data used to estimate the model parameters. When data indicates that C is 0 or negative, this modelling approach cannot be used to estimate the shift in optimum UEC indicated by consideration of SCC.

Based on the methodological steps outlined above, empirical data about appliance price, efficiency and UEC were used to statistically estimate the minimum LCC, the value of UEC at the LCC minimum, and the curvature of the LCC vs. UEC function near the LCC

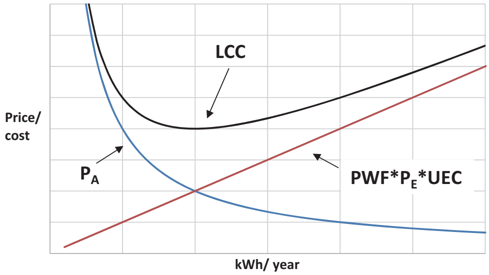

Fig. 1 Schematic illustration of LCC equation and determination of LCC minimum

[5] It is assumed that the effect of internalizing the climate externality on appliance prices does not systematically differ between efficiency classes. If, against this assumption, there is a systematic difference, it can be assumed to be small compared to difference in use-phase emissions, because the emissions caused in the production of a home appliance are in most cases small compared to indirect emissions from the use phase. For a thorough discussion of embodied emissions of products under the Ecodesign Directive, see Scott et al., 2017

[6] The quadratic approximation of the LCC near the minimum follows from Taylor's theorem in mathematics since at a minimum, a function has no first derivative. When both the second and third derivatives of the LCC function are non-zero, the quadratic approximation is likely to be valid when $|\mathrm{UEC}\text{-}\mathrm{UEC}_{\min}| << |\mathrm{LCC}''/\mathrm{LCC}'''|$ where LCC″ and LCC ‴ are the second and third derivatives of LCC with respect to UEC respectively.

minimum for tumble dryers and dishwashers[7] on the UK market. In this estimation, market data were used to estimate a reference line that provided market average UEC vs. appliance capacity (i.e. the number of place settings for dishwashers, and the kilogrammes of clothes drying capacity for tumble dryers). The LCC was then examined relative to this market average energy use and was fit to a quadratic function of energy use relative to the reference. Finally, Eq. 6 was used to estimate the shift in energy use implied by a MEPS policy that is based on LCC optimisation and considers SCC.

Estimating carbon prices to achieve the same energy efficiency improvements as MEPS

Equation 6 above addresses the question: for a given SCC, what is the corresponding shift in UEC? Alternatively, the reverse question can be asked: for a given shift in UEC, what is the corresponding carbon price (CP) that can make a switch from a lower to a higher efficiency class economically beneficial? To answer that second question, the SCC in Eq. 3 is replaced with CP. Then, Eq. 3 is equalized for pairs of efficiency classes. Solving for CP results in the following:

$$\mathrm{CP} = \frac{\frac{P_{A++} - P_{A+}}{\mathrm{PWF}} + (\mathrm{UEC}_{++} - \mathrm{UEC}_{+}) \times P_E}{(\mathrm{UEC}_{+} - \mathrm{UEC}_{++}) \times \mathrm{EF}} \tag{7}$$

where ++ indicates the more efficient appliance class and + the less efficient appliance class in the pair.

When modelled with market data, solving Eq. 7 resulted in the switching price that is needed in the UK to provide an incentive for economically rational consumers to shift from an average appliance model in one efficiency class to an average model in a higher efficiency class.

Key assumptions and sensitivity analysis

Besides appliance data, further information for key assumptions of the LCC modelling was needed (see Table 2). In order to get the present value of future electricity costs (and savings), a real social discount rate of 3.5% was used, which is the UK Government recommendation for central government policy evaluations (HM Treasury 2013). The *real* discount rate was used because zero inflation of electricity prices was assumed over the lifetime of the analysed appliances. The electricity price of 0.14 GBP/kWh was the average price for a consumer in the first half of 2016.

For the modelling, the average CO_2 emission factor of the UK electricity mix in 2014 of 413 gCO_2/kWh was used, which was likely not only above the average emission factor that can be expected over the lifetime of the appliances but also below the current marginal emission factor of the mix. The average lifetime of the different appliances was taken from literature, which in turn was the basis for calculating the PWF with Eq. 2. The SCC estimate used for this study was 150 USD (120 GBP). A high SCC value was chosen for this study in order to cover the whole range from not internalising the climate externality ('pure' LCC approach) to being confident that it is fully internalized (LCC with SCC of USD 150).

This exploration of a whole range of SCC estimates indicates the sensitivity of results with respect to changes in the assumed SCC. In order to further test the robustness of key results, additional sensitivity analyses were conducted. This included a variation in electricity price, emission factor and PWF by ± 50%. Most current electricity prices in the EU are included in the resulting interval of 0.7 to 0.21 GBP/kWh, and, with few exceptions, the average emission factors of most EU electricity mixes are contained in the interval 207 to 620 gCO_2/kWh (IEA, 2017). For the PWFs of the four appliances, the lower halves of the respective intervals seem to be more relevant, as on appliance markets, consumer discount rates have been found to frequently and significantly exceed market discount rates (Schleich et al., 2016; Wada et al., 2012), reflecting a lower present worth of future energy costs.

Results

Internalising the climate externality in the LCC of home appliances

Figure 2 shows average price and LCC curves for the four analysed appliances and their respective efficiency classes on the UK market in 2016. The general appliance price trend is clear: the lower the annual energy use of an appliance, the higher its price (with the exception of televisions, which will be further discussed below). In

[7] For refrigerators and televisions, the estimated LCC function did not allow for applying the outlined approach because for televisions, the minimum was outside the empirical data range and for refrigerators, the constant *C*, describing the curvature at the LCC minimum, was negative.

Table 2 Key assumptions used in LCC modelling

Description	Value	Source/comment
Discount rate (real)	3.5%	HM Treasury (2013)
Lifetime dishwasher (PWF in brackets)	12.5 years (10)	Boyano Larriba et al. (2017)
Lifetime refrigerator (PWF in brackets)	16 years (12)	VHK and ARMINES (2016)
Lifetime television (PWF in brackets)	7 years (6)	Stobbe (2007)
Lifetime tumble dryer (PWF in brackets)	13 years (10)	Lefèvre (2009)
Electricity price	0.14 GBP/kWh	Department for Business, Energy & Department for Business, Energy, and Industrial Strategy (2016)
Emission factor of the electricity mix	413 gCO_2/kWh	IEA (2016b)
Social cost of carbon	150 USD/tCO_2	High-end assumption based on literature review
Exchange rate USD–GBP	1.25	Approximate market exchange rate in early 2017

contrast, LCC trends vary: the least efficient refrigerators and dishwashers also have the lowest LCC; television models in the least efficient class have the highest LCC; and tumble dryers have the lowest LCC in efficiency class A+ and a higher LCC both for more efficient and less efficient models.

If the SCC is accounted for, the LCC-ranking of efficiency classes is affected only to a small degree.

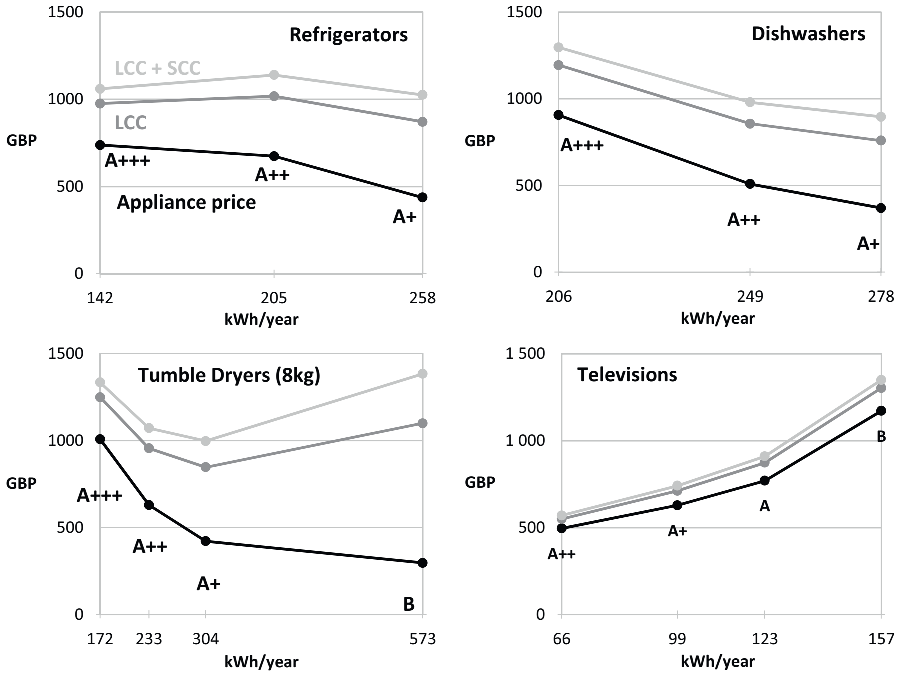

Fig. 2 LCC curves for appliances in the UK, with and without SCC

The main change is that for tumble dryers, the highest LCC moves from the average model in class A+++ to B, which is a strong argument in favour of mandating a standard higher than B. A higher standard may also be justified for refrigerators, for which the LCC difference between efficiency classes becomes very small if SCC is accounted for. Moreover, the individual model with the lowest LCC in each appliance category might well be a more efficient one if SCC is included, which means that the technological potential for cost-effective abatement through MEPS is likely higher than indicated by the aggregated data presented in Fig. 2.

While the inclusion of SCC only somewhat alters the LCC-ranking of appliance classes, it clearly increases the *level* of LCC. The space between the LCC curve and the LCC curve with SCC is where LCC would be located for lower SCC estimates between 0 and 150 USD per ton of CO_2. Furthermore, the area between the curves illustrates where LCC estimates would be located for lower emission factors between 0 and 413 g/kWh (at a constant SCC of 150 USD). Higher emission factors than 413 g/kWh, higher electricity prices than 0.14 GBP/kWh, and a SCC even higher than 150 USD would all result in an upward rotation of the LCC curve, which means a smaller increase of LCC for very efficient appliances and a larger increase for inefficient appliances.

In examining the special case of TVs, the relationship between energy efficiency and product prices appears to be reversed. One explanation for this counter-intuitive finding could be that accounting for screen size was not sufficient to isolate the marginal cost of energy efficiency. If, however, the analytical approach is refined and other product features are accounted for, such as NFC, smart television, screen type, screen resolution and number of tuners, the general trend still holds and price decreases with increasing efficiency.[8] This finding supports a previous study that found the same trend (Siderius, 2013).

Finally, Fig. 2 provides some information about the energy savings (and related mitigation) potential of energy-efficient technologies that are already on the market. A rough indication of this potential can be obtained from comparing average UEC of the median efficiency class (based on the sales data presented in Table 1) to average UEC of the most efficient class. The following differences can be observed: UEC is 45% (116 kWh/year) lower for A+++ refrigerators compared to A+ models, 26% (72 kWh/year) lower for A+++ dishwashers compared to A+ models, 70% (401 kWh/year) lower for A+++ tumble dryers compared to B models and 46% (57 kWh/year) lower for A++ TVs compared to A models. These indicative estimates of energy savings potentials are largely in line with previous research, suggesting that the energy reduction potential of EU product regulation by 2030 is 60% for refrigerators, 33% for dishwashers, 25% for tumble dryers and 64% for televisions (Kemna and Wierda, 2015).

The climate externality's impact on the LCC optimum of home appliances

While Fig. 2 displays the situation for the averages of different energy efficiency classes, a refined estimate of the shift in optimum UEC due to consideration of SCC can be obtained by using Eq. 6. Figure 3 illustrates the estimated LCC curvature close to the LCC minimum for dishwashers and tumble dryers. This quadratic function fit provides an estimate of the LCC minimum and the curvature of the minimum. For dishwashers, the estimated curvature is GBP 0.10 per $(\text{kWh/year})^2$ and for tumble dryers, it is GBP 0.01 per $(\text{kWh/year})^2$.

Using these curvature values, a SCC of 150 USD/ton implies a shift in optimum UEC of 25.4 kWh/year for tumble dryers and 2.5 kWh/year for dishwashers, which in relative terms represents shifts of 7 and 1% respectively. This means that in the context of LCC-optimized MEPS, a relatively small shift in MEPS can already account for the SCC. As can be seen in Eq. 6, this shift is fully proportional to the respective SCC, PWF and emission factor. If, for example, the SCC is doubled, which roughly reflects the shadow carbon prices of 1.5 °C-consistent scenarios in 2030, also the shift in optimum UEC is doubled, in this case to 2% for dishwashers and to 16% for tumble dryers.

The LCC optima that are seen in Fig. 3 were determined by fitting a curve to market data. This approach can be criticized for not capturing all the information there is in the distribution of individual models. There are, for instance, models on the market that have a UEC

[8] The specific average appliance prices and UEC of televisions on the UK market are the following if the more extensive regression model is applied: A++ (GBP 451; 65 kWh/year), A+ (671; 97), A (707; 121) and B (907; 154).

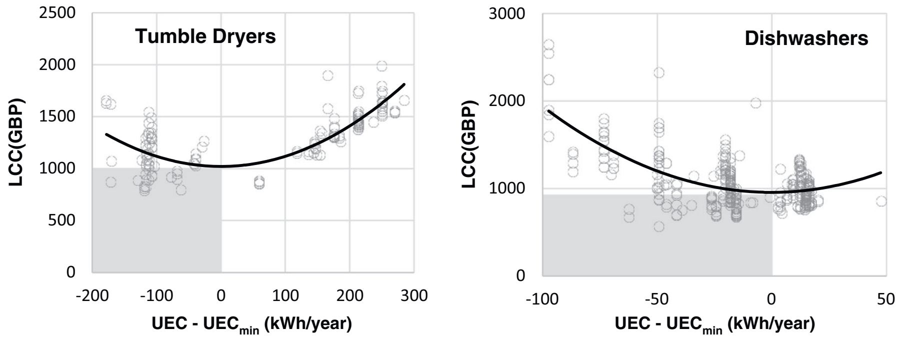

Fig. 3 Curvature of LCC minimum estimated from market data for tumble dryers and dishwashers

below UEC_{min} and LCC below LCC_{min} (see grey boxes in Fig. 3), which implies that the technological potential for cost-effective energy efficiency improvements of home appliances goes even beyond UEC_{min}.

A carbon price that sets the same incentive as a progressive MEPS

In Fig. 4 below, the perspective taken in Fig. 3 is turned around and a regulated shift one or two efficiency classes up (e.g. from A+ to A++) is compared to the carbon price that would be needed to incentivize the same shift. Note that this approach is not aimed at projecting actual consumer response but at identifying switching prices between average models of different efficiency classes on the UK market for an average consumer who considers full LCC when purchasing appliances. In order to see how assumptions about the electricity grid (emissions factor and electricity price) product lifetimes (as an element of PWF) and consumers' rationality (discount rate as part of the PWF) affect switching prices, Fig. 4 also includes the results of a sensitivity analysis.

The carbon prices displayed in Fig. 4 reveal several clear trends. First, for all appliances except televisions, carbon prices would have to be much higher than they are today, and even higher than the SCC estimate of USD 150 per ton of CO_2, in order to incentivize a switch between efficiency classes. For televisions, on the other hand, no carbon price is needed and lower LCC should already be incentive enough to purchase a model from the most efficient appliance class.

Second, the graphs depicting changes in the emission factor clearly show that the required carbon prices react exponentially. As electricity grids get decarbonized, it gets more and more difficult to incentivize the purchase of more efficient appliances by means of carbon pricing, because the carbon footprint of using an appliance is reduced over its anticipated lifetime. In the extreme case of countries like Norway and Sweden with CO_2 emission factors below 10 g/kWh (IEA, 2017), the required carbon price to incentivize a switch between efficiency classes approaches infinity. This illustrates that carbon pricing may only be a useful instrument to promote the purchase of efficient home appliances in a sufficiently 'dirty' electricity grid. For grids that are largely decarbonized, MEPS can still move appliance markets towards more efficiency, but due to low emission factors, energy savings translate, at best, into marginal CO_2 emissions reductions.

Third, and irrespective of the emission factor, Fig. 4 shows that increasing electricity prices bring the LCC of different efficiency classes closer together so that not such a high carbon price is needed anymore to incentivize a switch to the more efficient model class. But the figure also shows that electricity pricing alone will not be sufficient and significant carbon prices are needed. For the UK data, it is only the shift from A+ refrigerators to A++ refrigerators that could potentially be incentivized by a 50% increase of electricity prices alone.

Finally, and most importantly, Fig. 4 clearly shows that a departure from the unrealistic assumption that consumers fully consider LCC requires exponentially higher carbon prices in order to incentivize a shift to a higher efficiency class. While in welfare policy and SCC estimation it is most suitable to apply a social discount rate (3.5% in this study), implicit discount rates of consumers are typically around 20% or higher (Wada et al., 2012). The high discount rates reflect behavioural

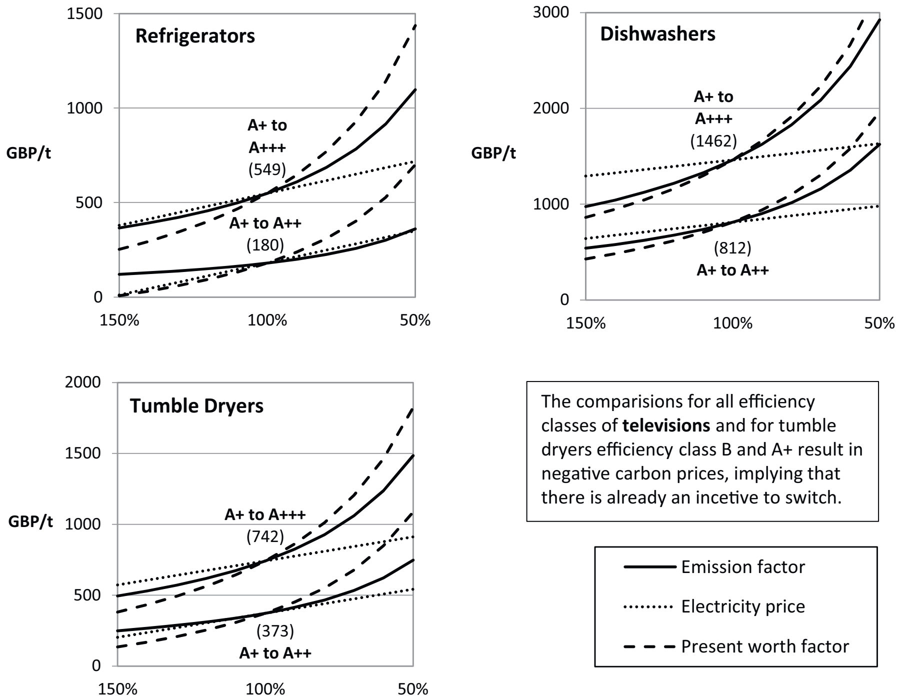

Fig. 4 Sensitivity analysis of switching prices for energy-efficient appliances. Displayed carbon prices represent a shift of MEPS by one or two efficiency classes. Sensitivity of results was tested with respect to changes in emission factor (100% = 413 gCO_2/kWh), electricity price (100% = 0.14 GBP/kWh) and PWF (100% = 10 for dishwashers and tumble dryers and 12 for refrigerators). Note that the analysis does not include the SCC

failures such as inattention, myopia, reference-dependent preferences and bounded rationality (Gerarden et al. 2017; Schleich et al., 2016). At a discount rate of 20% the PWF is roughly cut to half, which—as can be seen in Fig. 4—results in a steep increase of switching prices. If, on the other hand, some consumers expect electricity price increases that go beyond regular inflation, their discount rate might be lower, which results in a higher PWF and lower switching prices. Average consumer discount rates of 20% and more, however, indicate that such consumers are the exception.

Discussion

Methodological contributions and limitations

The analysis of the UK market for four electric home appliances has shown that the SCC (or a shadow carbon price) can easily be included in the modelling of LCC optima. This simple methodological approach has the potential to strengthen the effectiveness of MEPS as climate mitigation policy instruments. However, the approach to include SCC in LCC modelling has the same limitations as the LCC approach itself. It works well if there is a strong association between appliance prices and UEC, i.e. if higher prices imply more efficient products (Siderius, 2013). If this association is weak or even reversed (as could be observed in the case of televisions), LCC optimisation is of little use and other approaches should be used, e.g. a simple rule that a MEPS is set at the bottom end of the best performing quintile on the market, an approach known as ‘top-runner’ approach (Siderius, 2014).

A challenge of setting MEPS based on LCC optimisation is that this is typically a retrospective approach which has difficulties accounting for experience curve effects (Siderius, 2013). While multiple snapshots over time can be analysed to show trends as part of a real-time LCC methodology, future product improvements are difficult to predict. Also, the approach of looking at whole efficiency classes and fitting data to average trend curves is conservative by nature, because there are always models that outperform the average even in the most efficient class. In this way, technically feasible and cost-effective efficiency improvements are potentially hidden in aggregated data and behind a static retrospective modelling approach.

Another potential limitation of the LCC optimisation method used in this study is that it assumes a quadratic function to identify the LCC minima. There are several other functional forms that could be used to model minima, and an initial analysis showed that including cubic terms in the fitting function changes the SCC-induced shift of UEC at the LCC minimum by about 10%. Further research is needed in order to obtain more robust model specification variability. Still the main conclusion of this research seems to hold independently of the fitting function. If the LCC minimum is well defined, including SCC makes a moderate change in the electricity price, which results in a minor shift of the UEC associated with minimum LCC.

The modelling of switching prices that correspond to stringent MEPS is the second methodological contribution of this paper. In this study, the estimation of switching prices illustrates well the limitations that carbon pricing can have to incentivize investments for energy-efficient technology, particularly if discount rates above the market discount rate are assumed. While currently even the most progressive carbon pricing schemes are unlikely to have a significant steering effect, the overall mitigation effect of carbon pricing interventions also depends on the revenue use, which, in theory, can be fully targeted at climate change mitigation.[9] Revenue use was not considered in this analysis, as in practice, earmarking revenues for climate change mitigation is not yet a priority of existing carbon pricing schemes (Stiglitz et al., 2017). In addition to MEPS and carbon pricing schemes, it should be noted that there are further (combinations of) policy instruments that may address market failures and behavioural anomalies in an effective way but were outside this study's scope (for example product labels, subsidies and rebates).

Finally, the sensitivity analysis illustrates that what is optimal may differ between countries with different electricity prices and emissions factors and even between individual consumers who can discount future operating costs differently and value energy efficiency at different levels. While the analysis considered variation of the average consumer, consumer heterogeneity was not explicitly modelled.[10] Accordingly, stringent MEPS may not be welfare-enhancing for all consumers, and some consumers may be excluded from the market because of higher up-front costs of the most energy-efficient products. In such a case, consumer subsidies, tax breaks or other policies aimed at certain consumer groups may be a useful complement to MEPS. While MEPS appear to have been contentious in only limited cases, for example, the MEPS effectively 'banning' incandescent lightbulbs (see e.g. Frondel and Lohmann, 2011 and Sandahl et al., 2006) and the MEPS for vacuum cleaners in the UK (Barford and Dalhammar, 2015), these highlight the need to also ensure that there are no significant trade-offs with product quality and more stringent MEPS (though other research has found generally that product quality improves—see Brucal and Roberts, 2017).

Implications for 1.5 °C-consistent energy efficiency policy

Energy-economy modelling of climate scenarios has shown that delaying mitigation and increasing energy consumption render the 1.5 °C target unfeasible and the 2 °C target more costly (Clarke et al. 2014; Guivarch and Rogelj, 2017; Rogelj et al., 2015b; Waisman, 2017). Against this background, the policy mix for demand-side technologies needs to be both ambitious and quickly implemented. The results of this research are discussed regarding the potential of MEPS to function as a relevant climate policy towards the 1.5 °C target. In this discussion, special attention is given to the internalisation of SCC, the consistency of MEPS' effectiveness with 1.5 °C pathways, as well as their short- and

[9] It should be acknowledged here that the important issue of revenue use was added to the discussion after an anonymous reviewer highlighted its relevance.

[10] Similarly, standards can be perceived as a costly burden by some manufacturers but not by others. EU appliance manufacturers (BSH et al. 2012) and Swedish industry (Jönbrink and Melin 2008) seem to accept or even be in favour of more stringent product regulation. Such regulation can be a competitive advantage and experienced firms have learned they can comply with it at reasonable costs. However, in countries dominated by low-cost producers, the perceptions may be different and manufacturers may see stringent EU regulations as cost-increasing and as market barriers.

long-term economic efficiency. As MEPS can only be one element of a wider climate policy portfolio, the role of MEPS in the climate policy mix is also discussed.

The results from LCC modelling of four home appliances show that *a significant climate externality can be captured by MEPS* that are not much more stringent than current levels. At minimum, such low-hanging fruits should not be left hanging, even if the additional mitigation resulting from these adjustments cannot be expected to be a sufficient contribution towards reaching the 1.5 °C target. However, while incorporating SCC will internalize an externality, stringent MEPS that go beyond this are needed to drive significant CO_2 reductions from home appliances.

The findings also imply that much more stringent MEPS are, in principle, able to achieve a mitigation effect that is consistent with the 1.5 °C target. For the UK, deep decarbonisation scenarios consistent with the 2 °C target imply reductions of final energy consumption by about 10% in 2030, but to move towards 1.5 °C, further (not quantified) reductions in energy demand are needed (Pye et al. 2015). Our analysis and previous studies (Kemna and Wierda 2015; VHK 2016) have shown that energy savings from home appliances of 25% and more are feasible if the whole market for these appliances in the UK was shifted by stringent MEPS to the average performance of the currently highest efficiency class. If implemented swiftly, the realisation of emissions abatement driven by stringent MEPS still depends on the lifetimes (and associated stock turnovers) of the respective appliances, which in this study range from 7 years (televisions) to 16 years (refrigerators). Moreover, the emission factors of electricity grids determine the specific abatement associated with energy savings. In most countries, including the UK, electricity grids are still ‘sufficiently dirty’ (IEA 2017), implying a high abatement potential of efficiency improvements in electric appliances (Dietz et al. 2009; Letschert et al. 2013). In summary, the combination of current savings potential, stock turnover duration and emission factor appears to be largely consistent with the UK pathway for reaching the 1.5 °C target.[11]

Moreover, the mitigation effect of national or regional MEPS can be expected to go beyond the boundaries of the regulated markets. US and EU MEPS are effectively adopted by commercial actors in other jurisdictions due to the size of their markets (Bradford 2012) and have been the blueprint for MEPS in other countries (Molenbroek et al. 2015). This is important since it has been shown that climate mitigation efforts of front-running countries and regions (such as the EU) have significant climate benefits, but that they are not sufficient to limit global warming to 1.5 °C (Kriegler et al. 2015), and a wider coverage of comprehensive climate mitigation efforts is needed in the short term.

While stringent MEPS can be highly effective and this effect may spillover to unregulated markets, their short-run economic efficiency appears to be low. This is evidenced from the current switching prices for ambitious energy efficiency improvements found in this study, which range from 180 GBP/t CO_2 (switch from average A+ to A++ refrigerator) to 1460 GBP/t CO_2 (switch from average A+ to A+++ dishwasher). As these switching prices are higher than shadow carbon prices for reaching the 1.5 °C target (of about USD 200–300 per ton), in most cases, stringent MEPS do not appear to be cost-effective on the short run. The exception is stringent MEPS for televisions, which appear to be highly cost-effective already, as LCC of televisions are lowest in the highest efficiency class.

While high switching prices indicate high abatement costs for several appliances, it has been argued that technological roadmaps and policies should not only be based on *current* marginal costs but also on cost dynamics over time (Stiglitz et al. 2017, p. 29), a perspective also referred to as ‘dynamic efficiency’ (del Río González 2008). From a dynamic perspective, the cost-effectiveness of measures to reach a short-term target then also depends on long-term targets. For a demanding climate objective, such as the 1.5 °C target, the optimal strategy might well be to quickly and simultaneously implement measures with a wide range of marginal abatement costs (Vogt-Schilb and Hallegatte 2014). Moreover, recent evidence suggests that high switching costs for energy efficiency improvements of home appliances disappear (or are even reversed) if a dynamic perspective is taken, as prices for efficient home appliances have declined in relation to performance over time, especially when more stringent standards were enforced (Brucal and Roberts 2017; Van Buskirk et al. 2014). This is in line with previous research showing that the average prices of highly efficient tumble dryers and refrigerators fell with increasing

[11] The energy saving effect of efficiency improvements can be limited by rebound effects, which are estimated to reduce energy savings by 5–15% (Letschert et al. 2013; Sorrell 2007) or enhanced by technological progress, which—to a certain extent—cancel each other out and were not considered in this analysis.

market shares as learning and scale effects factored in (Siderius 2013).

Both their potential for dynamic efficiency and high effectiveness are arguments in favour of using stringent MEPS to complement carbon pricing instruments, but it has been argued that MEPS are not economically efficient, in particular in combination with ETS schemes (Böhringer et al. 2016). However, even from a purely economic perspective, the departure from the first-best policy approach of global comprehensive carbon pricing (Goulder and Parry 2008) does not have to lead to large efficiency losses. For the 2 °C target, energy-economy modelling has shown that a mix of modest carbon pricing with low-carbon energy technology policies can be nearly as efficient as global, comprehensive carbon pricing at a high price level (Bertram et al., 2015b). Considering that likely 2 °C scenarios make nearly comprehensive use of all supply side mitigation measures, and considering further that additional demand side measures are crucial for the 1.5 °C target (Rogelj et al., 2015a), it is likely that energy efficiency technology policy, such as MEPS, compromises the cost-effectiveness of carbon pricing instruments even less in a 1.5 °C context and—as outlined above—may be dynamically efficient. Moreover, if carbon pricing is implemented via emissions trading schemes, such as the EU ETS, emission reductions that are triggered by MEPS can be accounted for by adjusting the emissions cap, so that the carbon price incentive for other sectors is not diluted (Hood, 2013; Richstein et al., 2015; Sonnenschein, 2016). In practice, however, the predictability about impacts of any kind of energy (efficiency) regulation is still limited, so that there is a general need for a flexible adjustment mechanism of the supply with emission allowances (LBST et al. 2013).

Conclusions

Modelling of climate change scenarios has shown that radical energy efficiency improvements have to be realized immediately in order to keep alive any possibility to limit global warming to 1.5 °C until the end of the century. In this context, energy efficiency policy cannot afford to exclusively rely on weak carbon or energy price signals and uncertain market and behavioural response. This study has shown that even the high estimates of carbon prices required to limit global warming to 1.5 °C will not be enough to move markets for several electric home appliances towards BAT. Setting more stringent mandatory standards, on the other hand, can be seen as a way to force markets for home appliances towards more efficiency and realize their emissions abatement potential. In order to make use of the full abatement potential, stringent MEPS have to go beyond the incorporation of SCC in the underlying LCC modelling. If the time perspective is confined to the present, stringent MEPS do not appear to be the most economically efficient abatement option. If, however, technology pathways for reaching the 1.5 °C target are considered, and it is taken into consideration that prices for highly energy-efficient appliances have dropped quickly in the past, stringent MEPS not only are effective but also promise to be a cost-effective abatement policy.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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Paper IV

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Is one carbon price enough? Assessing the effects of payment vehicle choice on willingness to pay in Sweden

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Sweden

ABSTRACT

Existing knowledge suggests that people's willingness to pay (WTP) for climate change mitigation depends not only on personal characteristics but also on the payment vehicle (PV) that is used to elicit WTP. The aim of this research is to investigate policy-relevant differences in WTP between different PVs to support the design of carbon pricing mechanisms. The novelty of this contingent valuation study is the randomized use of four different PVs for the same sample (n = 500), in order to isolate effects of PV choice from effects driven by differences in study context and sampling. The results show that mean WTP differs between PVs. At about EUR 55 per tonne CO_2 it is highest for a climate surcharge on short distance flights, followed by the climate surcharge on long distance flights (EUR 36), the climate surcharge on fuels (EUR 32) and voluntary offsetting (EUR 14). Statistical tests show that for almost all pairs of PVs the differential effect of PV choice is significant. Moreover, the results illustrate that WTP means are sensitive to changes in the assumed carbon intensity of the respective energy-consuming activities (in particular for air travel). In all, the differential effects of PV choice suggest that a uniform carbon price is inadequate and prioritisation and differentiation are needed in policymaking.

1. Introduction

In order to stay within the limits of global warming outlined in the Paris Agreement [1], low-carbon technologies and mitigation practices in various areas of production and consumption have to be implemented as soon as possible [2–4]. From a consumers' perspective, the mitigation potentials and costs associated with decarbonization vary across areas of consumption, such as travelling, electricity consumption or the consumption of food [5]. Whether or not people are willing to change their consumption behaviour or support climate mitigation policies not only depends on costs, but also on contextual influences (sociodemographic factors, life situation, policies), personal influences (information, attitudes, awareness, beliefs) attributes of the required behavioural change (comfort or service level), and not least on the value people attach to the associated environmental benefits [6,7]. A common way to investigate the latter, the perceived environmental benefits of consumption decisions, is to elicit people's willingness to pay (WTP) for these non-market values [8].

Several studies have elicited people's WTP for mitigating climate change at a household or individual level, but they vary in their geographical scope and in the consumption area they cover. In the context of flying, WTP for climate change mitigation has been elicited both per ton of CO_2 and per flight, and results suggest that there is a significant positive WTP which differs between people of different country-origins and incomes [9,10]. In the context of WTP for using green electricity as a means of climate change mitigation, studies show that WTP is heavily dependent on respondents characteristics, such as age, education and wealth [11], the payment mechanism [12], survey design [13], and even on the actor that generates the green electricity [12,14]. For national climate change policies, WTP has been shown to depend on the type of policy instrument, such as a carbon tax or climate change regulation [15], and the question format [16]. Some recent experimental studies have investigated the WTP for the retirement of EU ETS allowances (EUA) as a way to offset personal emissions, concluding that incentivized studies may lead to lower WTP than survey studies [17–19]. Overall, contingent valuation of climate change mitigation has been practiced in various areas of consumption, has applied a variety of payment mechanisms, has made use of different WTP elicitation formats and has been applied to a diverse set of samples. Little research has been done, however, in *combining* different areas of consumption

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and payment mechanisms in a single contingent valuation study, which this paper addresses.

When it comes to estimates, predictably, there is a large variation of WTP for climate change mitigation across the aforementioned studies. Mean WTP per tonne of CO_2 varies between five and several hundred of Euros (EUR), and if measured as an annual contribution, WTP for climate change mitigation is found to vary between EUR 25 and 300 [20]. Review studies that compare WTP estimates and their drivers suggest that WTP depends on the socio-economics of the sample, the specific local or regional circumstances, the elicitation format (stated or revealed), and the payment vehicle (PV) being used to elicit WTP, such as fuels taxes or voluntary emissions offsets [17,20]. As studies generally differ with respect to more than one of these aspects, it becomes difficult to isolate the substantive effect, which the choice and design of PV have on WTP, from the effects of study context, sampling and choice of elicitation method. This makes it problematic to derive policy recommendations from potential differential effects of PV choice.

There are several disciplinary approaches to investigate differential effects of PV choice in particular, and decision-making in the context of climate mitigation and sustainable energy use in general, including conventional and behavioural economics, technology diffusion, social psychology or sociology [21]. From a conventional economic perspective, different WTP values for different PVs can be seen as expressions of preferences for behaviours that are associated with the respective PV, e.g. the avoidance or changing costs that consumers associate with higher fuel taxes. This study, however, takes a behavioural economic perspective that views preferences as context-dependent [22]. Hence, the objective is not the *observation* of climate mitigation preferences per se, but the *investigation* of climate mitigation preferences in different contexts, with the aim to generate policy-relevant practical knowledge. This research approach reflects a critical realist epistemology, where the focus is on *explaining* (a certain WTP) rather than predicting or interpreting [23]. While it seems unlikely to find an unbiased PV that only elicits WTP for an environmental good and nothing else [24], it is important to explain differences across PVs and their framings in order to derive policy implications.

The sparse existing work that explicitly touches upon the effect of PV choice on WTP for climate change mitigation provides mixed evidence. Some studies have found (framings of) PVs that appear to increase WTP: making the PV the default option [25], applying a collective, coercive PV instead of a voluntary PV [12,26,27], leaving revenue use for green energy provision to the private sector instead of the public sector [12], eliciting WTP with a PV that implies low absolute costs [28], and applying a PV in the context of high environmental impact [28]. The reviewed literature strongly suggests that PV choice and contexts matter, but further research is needed to get more robust evidence about the direction and strength of the effect of this choice for different PVs and their framings.

Given this research gap, the objective of this study is to investigate policy-relevant differences in WTP between different PVs to better inform the design of consumption-based carbon pricing schemes and complementary mitigation policies. A key question is whether a uniform carbon pricing mechanism (explicit or implicit) is justified, considering both the urgent need for effective mitigation and potential differences in WTP between PVs. From a methodological point of view, our study uses one bidding structure in repeated elicitation of WTP in one sample in order to directly compare PVs and energy use domains and detect differential effects. Using consistent bidding-structures, WTP per tonne of CO_2 is elicited across four different PVs: two variations of an air ticket surcharge, a fuel surcharge and voluntary offsets.

Empirical data collection took place in Sweden, where knowledge about climate change is widespread and it is perceived as the single most serious problem that the world is facing [29]. Swedes already have experience with explicit and implicit carbon pricing mechanisms as both a carbon tax and a CO_2 based motor vehicle tax are in place – in addition to various other instruments to implicitly price carbon emissions, such as energy taxes and product standards. We argue that investigating WTP for climate change mitigation in Sweden has the potential to feed into the debate and design of additional carbon pricing mechanisms such as the air ticket tax that has been recently introduced [30]. WTP data in the Swedish context is so far limited to people's general WTP for offsetting emissions from flights [31], and to household's monthly WTP for mitigation in general [32], but data on WTP per tonne of CO_2 and on differential effects are still lacking.

Based on indications from previous studies, it is hypothesized that variation of PVs will result in significantly different WTP values. More specifically it is expected that WTP is higher for coercive than for voluntary PVs [12], that WTP is higher if the PV implies low absolute costs for the respondent [28], and that WTP is higher for emissions from air travel than from car travel, because car travel emissions are already taxed significantly higher than emissions from air travel. In order to research these hypotheses, the study applied the contingent valuation method (CVM) and statistical techniques to analyse WTP data (see Section 2). The results of the quantitative analysis and several robustness tests are presented and discussed in the context of previous findings and related to concepts from behavioural economics (Section 3). Behavioural economics was considered a suitable theoretical frame because it has the potential to provide explanations for bounded rationality, such as inconsistent valuation of the same good (i.e. climate change mitigation) by the same actor. The main policy implications of the study are outlined and key conclusions are drawn (Section 4).

2. Research design and methods

2.1. A contingent valuation approach

CVM was used in this study to investigate the economic value people attach to climate change mitigation. This method was mainly chosen because there is no functioning market for individual CO_2 emissions reductions. Reflecting the challenges with revealed preference approaches, CVM has the advantage that it facilitates representative sampling, allows for within-subject comparisons, and enables the set-up of a policy-relevant and realistic choice context. Alternatively, choice experiments could have been conducted, which are used to identify trade-offs between attributes of goods or also policies such as carbon pricing (see [33]). For this study CVM was preferred because of its 'dominance' in the context of climate change mitigation ([20], p. 13), which facilitates comparisons between this study and existing knowledge.

One major critique of CVM is that, as *stated* preference approach, it is subject to hypothetical and strategic biases. The hypothetical bias is the systematic overstatement of individuals' economic valuation of a good in hypothetical choice situations, and has been shown to be particularly strong for public goods [34,35]. Closely related, it has been argued that CVM values moral satisfaction rather than economic value [36]. A strategic bias, on the other hand, can occur when respondents systematically undervalue a good because they feel that study results may lead to policies (e.g. fees or taxes) that will be costly for them in the future [37]. There is a trade-off between reducing hypothetical bias and reducing strategic bias, as making the choice setting more realistic may trigger strategic behaviour and vice versa [37].

This study took the conservative approach to focus on the mitigation of hypothetical bias. First, the CVM survey (see details in Section 2.2 below) applied 'consequentiality design' ([38], p. 36) by reminding participants that results will be used to inform 'future policy development and design', and thus, might affect respondents' future utility.

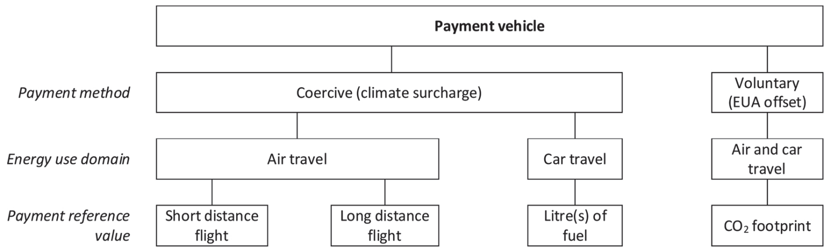

Fig. 1. Overview of payment vehicles (PV). The figure shows the four PVs that were used in this study to elicit WTP for climate change mitigation.

Second, respondents were told that they should not agree to expensive measures if they cannot afford them or there are more important things to spend their money on. Third, 'uncertain' was included as response option to valuation bids, which was recoded as a 'no' response to reduce potential upwards bias of maximum WTP. The only measure that was taken to reduce strategic bias was to explicitly remind participants that they are asked what they are personally willing to pay and not what they consider the right price level for a carbon pricing policy. Despite of all these measures, biases could not be fully precluded. However, remaining hypothetical or strategic biases are more relevant for the examination of absolute level(s) of WTP than for the analysis of differential effects between different PVs, the focus of this study.

To address these differential effects caused by PV choice and design, the core element of the CVM study, was the randomized use of four different PVs to elicit respondents' WTP (see Fig. 1). The PVs differed with respect to three dimensions: *i)* payment method (coercive or voluntary), *ii)* domain of energy use, and *iii)* reference value. There was one voluntary PV, namely offsets of transport emissions (both car and air travel) via the purchase and retirement of allowances of the EU emissions trading system (EUA). This PV was included to test the hypothesis that coercive and collective PVs result in higher WTP than voluntary ones. A mandatory climate surcharge was used as PV for both air and car travel in order to detect differences between energy use domains. Overall the focus was on modes of transport and their emissions in order to gain knowledge in an area of energy use where consumer decisions have a large impact (as recommended by Ref. [39]). Finally, a differentiation was made between short and long-distance flights in order to investigate whether there is a difference in WTP which may be explained by the 'low-cost hypothesis', i.e. WTP is higher for short-distance flights where the surcharge implies lower costs in absolute terms.

Table 1

Bid levels for the four payment vehicles. Note that bid values in rows are equivalent to each other. The conversion from the first to the other three columns is based on the respective carbon intensities, which are 2.5 $kgCO_2$ per litre of fuel, and 171 and 133 gCO_2 per person-kilometre (pkm) for short- and long-distance flights respectively.

EUA (SEK/ tCO_2)	Fuel surcharge (SEK/litre)	Flight surcharge (SEK/1750 km)	Flight surcharge (SEK/9000 km)
100	0.25	30	120
200	0.50	60	240
400	1.00	120	480
600	1.50	180	720
800	2.00	240	960
1000	2.50	300	1200

In order to elicit WTP for different PVs, this study combined a simple dichotomous choice question about general WTP with an iterative bidding process for all those respondents that were generally willing to pay (see Fig. 2). Giving respondents the option to agree to a PV in principle but reject (even the lowest) bids can help to reduce cognitive dissonance, as participants who are supportive of the PV as such, but find bids too high, get another option [40]. The initial bid was fixed. This might have resulted in an anchoring effect as this first bid might have been perceived by respondents as a recommended or commonly accepted central value [41]. However, the attempt was made to reduce the anchoring effect by following up the first bid with two more bids as laid out in Fig. 2. Again, any remaining bias might have affected absolute level(s) of WTP but not the differential effects.

Differential effects were likely unaffected by anchoring, as the bids

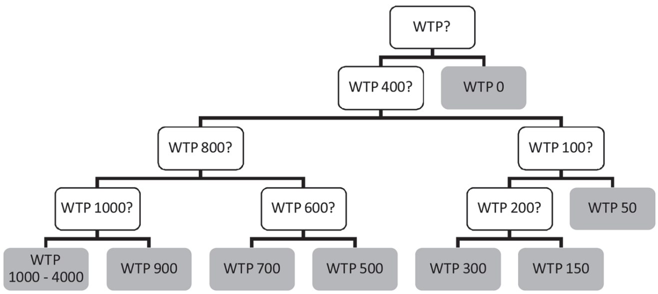

Fig. 2. Structure of the repeated bidding process. White boxes represent the bids offered in SEK/ t CO_2. The left path indicates acceptance of the previous bid, the right path rejection. Grey boxes indicate the WTP estimates obtained from repeated bidding (the mid-points of the respective intervals). Respondents who were willing to pay the equivalent of 1000 SEK/ tCO_2 could freely state their WTP between 1000 and 4000 SEK/ tCO_2.

offered in the survey were the same for all four PVs (see Table 1) and ranged from 100 to 1000 SEK/t CO_2 (10 to 100 EUR/tCO_2).[1] This range reflects both the experience from previous studies [9,18] and the price levels of current carbon pricing schemes, which range from as little as one EUR per tonne of CO_2 up to EUR 115 in the Swedish carbon tax scheme [42]. While corresponding to the same values in SEK/t, the bids were presented in different units, including SEK/l for the fuel surcharge and SEK/flight for short- and long-distance flights (see Table 1). Except for offsetting with EUAs, for which no unit transformation was needed, WTP was therefore not directly measured per tCO_2. For the fuel surcharge a transformation from litre of fuel to tCO_2 was carried out, for which a CO_2 intensity of 2.5 kg CO_2/l was applied. Diesel has a slightly higher CO_2 intensity (2.7 kg/l) than petrol (2.3 kg/l) [43] and the shares of diesel and petrol fuels in road transport in Sweden are comparable [44], so that the weighted average CO_2 intensity is about 2.5 kg/l.

For the air ticket surcharge, determining the CO_2 intensity was more challenging due to a lack of robust data. The values that were assumed and used in this study (i.e. 171 g/ pkm for a 1 750 km flight and 133 g/ pkm for a 9 000 km flight) were chosen for three reasons. First, they were approximately in the range of values provided by literature [45–47] and by the carbon emissions calculator of the International Civil Aviation Association (see online supplementary material for the specific carbon intensities). Second, within this range a high carbon intensity estimate was chosen so that WTP per ton of CO_2 was not overstated. Moreover, a high carbon intensity reflects at least partially that CO_2 equivalent (CO_{2}-eq) emissions from flying can be higher by a factor of about 1.9 than CO_2 [48], as also reflected in the study by Åkerman [45]. Third, the specific values that were chosen meant that round figures could be used for the bids presented in the survey (e.g. 9 000 km * 133 g/pkm * 100 SEK/t = 120 SEK), in order to reduce the cognitive load for respondents.

Each respondent was randomly exposed to all four PVs, which means that for every respondent WTP was elicited four times. The responses obtained from repeated bidding were treated as follows. A 'No' to the question about general WTP and a 'No' or 'Unsure' response to the lowest bid were coded as maximum WTP of zero. Otherwise, maximum WTP was determined at the highest accepted bid.

2.2. Survey design

The survey consisted of three blocks: respondent characteristics, elicitation of WTP and a short opinion poll.[2] The first block included questions about socio-demographics (age, gender, income, household size, and education), political view, and energy use behaviour. This initial set of questions was followed by the second block and the actual elicitation of WTP. Respondents were exposed to the PVs flight surcharge, fuel surcharge and EUA offsets in randomized order to avoid systematic bias due to fatigue, order or practice effects. The willingness to pay a surcharge for short- and long-distance flights was elicited one after the other but also in random order. For each PV, the elicitation of WTP started with a short info screen about the respective PV, followed by a question about respondents' general WTP, and in case of a positive reply, the iterative bidding process (see Fig. 2). In case of a negative reply, respondents were asked in a close-ended question why they were not willing to pay.

The third block of the survey, a short opinion poll, started by asking participants to rank-order the different PVs according to their preferences. In this question, voluntary offsetting with EUAs was explicitly framed as status quo in order to find out more clearly what the preferences for additional policy interventions are. The short opinion poll continued by asking participants to rank-order who is responsible for reducing CO_2 emissions from air travel and road transport. Then respondents were asked to rank-order different options what revenues from additional carbon pricing should be used for. At the end of the survey, respondents could freely share additional thoughts in a textbox (optional) and, invisible to the respondents, their total response time was recorded.

The design of the survey was tested with 40 volunteers and the test covered the following aspects: survey logic, response validation rules, language and understanding, and response time. The trials resulted in two major changes. First, the complex calculation of respondents' carbon footprint with an online tool was replaced by a simple self-assessment question. Second, the recording of participants' response time was integrated in the survey in order to screen out responses that were quicker than the fastest responses in a trial run (ca. 250 s), in which participants were asked to reply as quickly as possible but still read all the information.

2.3. Data collection and sampling

The online-survey was carried out in Sweden in January 2017. The statistical population of this study was the adult population of Sweden, aged 25–74 (6.2 million). The Swedish online panel 'panel.se' was used as a sampling frame. Panel.se had 67 500 active panellists who received points for their participation in surveys which can be exchanged for various non-cash benefits. From this frame, a random sample was drawn, which was representative with respect to age, gender and geographical region of the statistical population. The invitation of 1 507 panel members resulted in 500 completed surveys. A sample of 500 implies that mean WTP differences of 50 SEK/ tCO_2 can be detected with a statistical power of 0.8 (assuming a standard deviation of 400 SEK/ tCO_2 and a confidence level of 0.95). In addition to the 500 completed surveys, 48 incomplete ones were obtained and 126 were screened out. Leaving out the incomplete surveys, this implies a response rate of 42% (626 of 1 507). Among the screened responses 19 were excluded due to age, 89 as they were started after the quota of 500 was reached and 18 due to very short response times. In line with the existing literature, short response times suggest satisficing behaviour, which occurs in 5–30% of responses to online surveys [49] and has been shown to distort WTP estimates [50]. In this context, dismissing the 3% (18 of 626) fastest responses appears to be unproblematic; in contrast, it likely improved WTP estimates.

The socio-demographic overview of this study's population and the survey respondents is presented in Table 2. While the sampling frame (panel.se) was representative with respect to gender and age, a two-sided t-test showed that the final group of respondents was significantly older than the Swedish average ($t = 3.27$; $p < 0.01$). A proportions test, however, did not show a significant difference in the share of women ($z = -0.27$; $p = 0.77$). The median monthly income before tax of the Swedish population was in the median income interval of study respondents. The median income of the population was, however, based on a different age group (25–64) than the survey (25–74). The same holds true for household sizes, which were based on all ages for the whole population but not for the respondents (only 25–74). Therefore, the significant household size difference ($t = 4.52$; $p < 0.01$) between respondent group and the population mean of 2.2 is not a very robust finding. Taken together, the differences between population and group of respondents suggest that specific results have to be treated with care due to potential non-response bias [83].

[1] All monetary values in the survey were presented in Swedish Krona (SEK), the local currency of the research study. When Euro values are presented in this article, an exchange rate of 10 SEK/EUR was used, which is approximately the average exchange rate during the time of study in 2017/18 (source: xe.com).

[2] The complete survey is available in the online supplementary material. The survey data was used in a subsequent study, which focused on the air ticket surcharge and explanatory factors behind WTP for this PV [84].

Table 2
Socio-demographic overview of population end of 2016 and sample. The average number of people in a household refers to the whole Swedish population, not only the ones aged 25-74. For survey households with 'more than 5' people (n = 10) 6 people were assumed. The median income of the whole population assumes a tax rate of 30% and applies only to the age group.25–64.

	Population (age 25-74)	Respondents	SE	Data coding
Number	6 200 688	500		
Share of women	0.49	0.49	0.022	1 (female), 0 (other)
Age (mean)	48.48	50.54	0.631	25–74 (number in years)
Age (median)	48	51		
People per household (mean)	2.2	2.45	0.054	1 (1 person in household) to 6 (more than 5 people)
Median monthly income before tax (interval median in brackets)	29 100	20 000 – 29 999 (28 548)		1 (< 10 000 SEK), 2 (10 000–19 999 SEK) to 8 (> 70 000 SEK)

2.4. Statistical tests of effects of PV choice on WTP

In order to investigate the differential effects of PV choice and design on WTP for climate change mitigation several tests were computed.[3] To start with, various measures of centrality for the WTP distributions were obtained, including the arithmetic mean, the trimmed mean (5%), the median and the mean for respondents with WTP larger than zero. Focussing on the arithmetic means, two-sided paired *t*-tests and Wilcoxon signed-rank tests were conducted in order to investigate whether potential differences in mean WTP between PVs are statistically significant. The Wilcoxon signed-rank test is particularly suitable for data that is not normally distributed [51] such as WTP data. Effect sizes were calculated for the mean differences in order to go beyond statistical significance [52] and to get a better understanding of the differential effect of PV choice and design on WTP. Effect size computations included Cohen's d and r_t for the paired *t*-tests [53], where d is defined by dividing the difference between one mean (μ_1) and the other (μ_2) by the standard deviation of this difference ($\sigma_{1\text{-}2}$):

$$d = \frac{\mu_1 - \mu_2}{\sigma_{1-2}} \tag{1}$$

and r_t is the strength of association defined based on the *t*-score:

$$r_t = \sqrt{\frac{t^2}{t^2 + df}} \tag{2}$$

The effect size r_w measures strength of association based on the z-score from the Wilcoxon signed-rank tests [51] and is defined as:

$$r_w = \frac{z}{\sqrt{N}} \tag{3}$$

where N is the number of observations (or twice the sample size as for every respondent two WTP values are used in the comparison).

Besides tests of mean differences and effect size calculations, the robustness of mean WTP/tCO_2 to changes in the assumed carbon intensities for flights and fuel use was tested. For fuels, the test interval was $+/-10\%$ of a carbon intensity of 2.5 kg/l; and for long- and short-distance flights the test interval was $+/-50\%$ of 133 and 171 g CO_2/pkm respectively. The larger interval for air travel reflects the higher degree of uncertainty, which is due to scientific uncertainty about the global warming potential (GWP) of aircraft emissions [48] and due to differences in carbon intensity between air-craft models and routes [46].

3. Results and discussion

3.1. Bid acceptance

The survey revealed that approximately 70% of respondents were in principle willing to pay a surcharge on their air travel emissions, while only about 50% had a positive WTP for a surcharge on fuels, and less than 30% for offsetting with EUAs. For all four PVs the highest acceptance level can be found at the central (starting-) bid, which suggests an anchoring effect of WTP elicitation with fixed starting bids (see Fig. 3). However, the distributions of positive WTP responses above and below the central bid strongly differ between PVs. This, in turn, is a first indication that there are WTP differences between PVs. For short-distance flights 42.2% of respondents were willing to pay more than 150 SEK surcharge and only 14.4% less. The distribution is similar around the starting bid of 1.25 SEK/l fuel surcharge (21% willing to pay more, 16.6% less), while it is reversed for the long-distance flight starting bid of 600 SEK (17.4% more, 33% less) and the EUA offsetting starting bid of 500 SEK(7.6% more, 13.6% less).

3.2. The level of WTP for climate change mitigation

The central values for maximum WTP under the four different PVs give a first indication that the choice and design of PVs have an impact on the results that can be obtained (see Table 3). The mean WTP a climate surcharge on tickets for short-distance flights (WTP_{air_short}) is at 551 SEK/tCO_2 the highest, followed by the climate surcharge for long-distance flights (WTP_{air_long}) at 355 SEK/ tCO_2, the climate surcharge on fuels (WTP_{fuel}) at 317 SEK/tCO_2, and finally the offset with EUAs (WTP_{eua}) at 136 SEK/tCO_2. This order is the same for the more conservative estimates of median WTP and 5% trimmed means, but at a lower overall level as the 'fat tails' of the distribution have lower impact. As WTP values below zero were not allowed in the survey, standard deviations (SD) higher than the mean indicate a highly skewed distribution of WTP values.

The results of this study appear to be in the range of previous contingent valuations of climate change mitigation. Fig. 4 shows the levels of mean WTP for climate change mitigation that were found in this study in comparison to existing literature. Remarkably, mean WTP_{eua} is even below or at the same level of recent incentivized experiments [18,19], suggesting that the approaches to mitigate hypothetical bias have functioned. Mean WTP_{air_short} and WTP_{air_long} are at about the same level of previous studies from the Netherlands [9] and the UK [10], while other surveys from Australia [54] and Taiwan [55] found lower mean WTP.

About three quarters of global emissions covered by direct carbon pricing policies in 2017 were priced at below 10 EUR/tCO_2 [42], which is clearly below the WTP values found in this study. In Sweden, however, only parts of national emissions were priced at a comparably low level, and these were the emissions under the EU ETS with an allowance price of about 6 EUR/tCO_2 in 2017. In contrast, Sweden's carbon tax is the highest in the world (1 150 SEK/tCO_2); and for motor-vehicle fuels this carbon tax is even combined with an energy tax, resulting in an overall implicit carbon tax on fuels of more than 200 EUR/ tCO_2 [56]. In contrast, the explicit (and implicit) carbon tax on air travel was zero

[3] The survey data and analysis (stata.do-file) can be accessed here: https://osf.io/mp38d/?view_only=c508ad5753cd4ceabdf8c8ed8d437733.

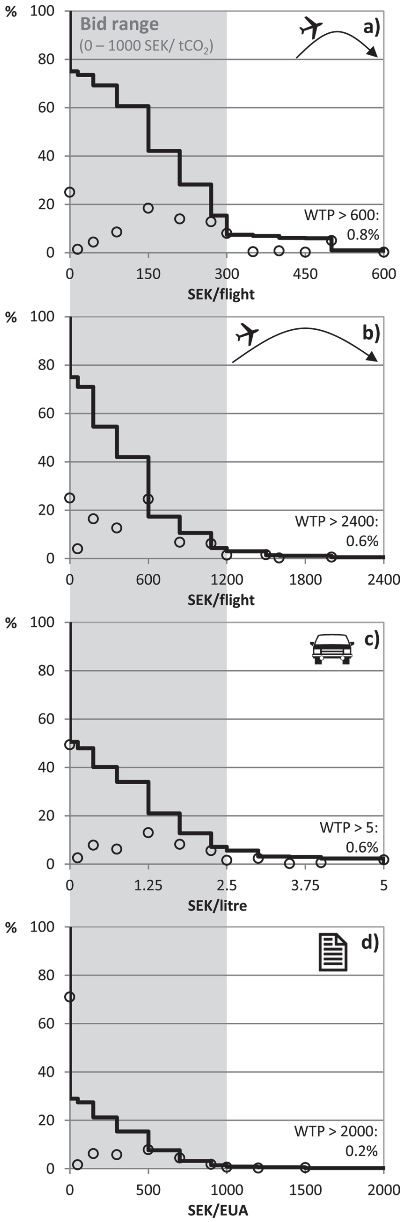

Fig. 3. Bid acceptance levels for the four payment vehicles: surcharges on a) short-distance flights, b) long-distance flights, c) motor-vehicle fuels, and d) purchase of EUAs. The circles indicate acceptance at given bid levels; the solid lines show cumulative acceptance. The grey area spans over the survey's bid range.

in 2017. Whether knowledge about explicit carbon prices in Sweden influenced responses directly is unclear.

On the other hand, an indirect influence of current carbon price level on responses is likely, because current carbon pricing levels informed the choice of response intervals for the WTP elicitation process (see Section 2.1) which in turn provided an anchor for respondents. If now absolute WTP levels of this study were to be compared to the same carbon prices that were used to design the elicitation process, there would be a serious risk of confirmation bias. Moreover, different base levels of carbon prices – 0 SEK on air travel versus > 2000 SEK/ tCO_2 on motor-vehicle fuels – further impede a straightforward comparison. Hence, absolute WTP values of this and similar studies should not be used directly in the process of determining carbon pricing rates. Mean differences, however, have the potential to provide policy-relevant insights.

3.3. Differences in mean WTP

Significant differences in mean WTP between PVs were identified, ranging from 39 SEK/ tCO_2 (WTP_{air_long} and WTP_{fuel}) to 415 SEK/tCO_2 (WTP_{air_short} and WTP_{eua}). Table 4 shows the respective test statistics for all possible pairs of PVs. However, neither the absolute size of mean differences, nor their significance level provide information how large the effect is of eliciting WTP for climate mitigation with one PV rather than the other. Effect sizes that take into consideration also variation of responses around the means can, in contrast, provide an indication about how sizable an effect is. In this study effect sizes differ between pairs of PVs and vary from around 0.1 (WTP_{fuel} and WTP_{air_long}) to about 0.7 (WTP_{air_short} and WTP_{eua}), which means that most effect sizes can be classified as '*medium*' [57,58]. A more substantial interpretation of these effect sizes, e.g. precise statements about the probability that a randomly chosen person has higher WTP_{air_short} than WTP_{eua}, cannot be made as WTP is not normally distributed [59].

The significant differences in mean WTP are largely consistent with respondents' ranking of carbon pricing alternatives, which was surveyed separately (see Fig. 5). The air ticket surcharge came out as the most favoured policy alternative, which was, surprisingly, even preferred to the status quo of voluntary offsetting.

The significant differences between WTP for climate change mitigation across different PVs relate to the existing literature in at least two ways. First, they support the previous finding that coercive (also referred to as 'collective') PVs, in this case the air ticket surcharge and fuel surcharge, are more preferred and elicit higher WTP values than voluntary PVs [12,26,27,60]. The mean WTP_{eua} was consistently lower than the mandatory PVs. Moreover, the air ticket surcharge was clearly preferred to EUA offsetting in the opinion survey. An established mechanism behind the difference between collective vs. voluntary contributions to a public good is conditional cooperation [61–64]. People tend to cooperate (i.e. contribute to a public good like stable climate) if they know, or at least have strong signals, that others also contribute. The clear majority of respondents who were not willing to pay for EUA offsets in principle stated that the main reason for this was that they did not believe in the impact of EUA offsets (202 of 355). For some respondents, this assessment might be driven by the belief that the EU ETS does not reduce emission effectively, for others by the expectation that very few people actually offset their emissions this way. If this is the case and respondents do not believe in the contribution of others, they do not contribute either if they are conditional co-operators. This can explain why WTP_{eua} is so much lower than WTP_{air_short}, WTP_{air_long} and WTP_{fuel}, all of which are coercive instruments that force people to contribute and avoid free-riding.

Second, the results support the low-cost hypothesis [28,65]. This hypothesis suggests that in a decision situation that implies lower

Table 3
Measures of centrality for maximum WTP. WTP values for climate change mitigation are shown in SEK/ tCO_2. Medians are computed both for treating WTP values as nominal and interval data.

	WTP_{air_short}	WTP_{air_long}	WTP_{fuel}	WTP_{eua}
Mean	551	355	317	136
SD	521	388	481	306
SE	23	17	22	14
Median (nominal)	500	300	50	0
Median (interval)	415	205	12	0
5% trimmed mean	504	317	253	94
Mean WTP > 0	(n = 375) 735	(n = 375) 474	(n = 253) 626	(n = 145) 469

absolute costs for a respondent, WTP for a good is higher. While the bids offered in the WTP elicitation with different PVs were the same per ton of CO_2, absolute bid values for WTP_{air_short} were, by construction, a factor of four lower than the bids for WTP_{air_long} (see Table 1). This difference in absolute bid values, fully proportional to the difference in CO_2 emissions, was likely the main driver behind a significantly higher mean WTP_{air_short}, thus implying higher WTP in low-cost payment situations. Knowing that there are viable alternatives with lower emissions to which they can switch might have led respondents to support a higher air ticket surcharge. While there is, indeed, a positive cross-elasticity between air and train travel demand in Sweden [66], the effect of available alternatives on WTP_{air_short} can be assumed to be small as the example short-distance flights were around 1 500 km long, a distance presumably few respondents would travel by train or bus in reality.

3.4. Potential framing effects

This study found clear differential effects triggered by the choice of PV. There are, however, further framing effects that likely influenced the results but were not controlled for systematically. First, the validity of the construct (i.e. PVs as a way to measure WTP for climate change mitigation) deserves attention. Despite careful framing of the PVs as measures to reduce CO_2 emissions, it is possible that respondents stated their current willingness to change their payment for an energy/ transport service, instead of their WTP for reducing CO_2 emissions *per se*. Moreover, there is some evidence that respondents have considered the amount of fuel taxes and even motor vehicle taxes that already exist in Sweden when eliciting their WTP. In fact, of those respondents who were not willing to pay a fuel surcharge in principle, 62% (153 of 247) stated that this was due to existing taxes on fuels and motor vehicles.

This evidence, that respondents assess PVs in the context of existing policy instruments, relates to another potential framing effect, namely strategic undervaluation. Carbon taxation and especially air ticket taxation were an important policy issue in Sweden at the time of data

Table 4
Comparison of WTP means. Mean differences (in SEK/ tCO_2) were analysed in paired *t*-tests (t) and Wilcoxon signed-rank tests (z), and effect sizes were computed (d, r_t and r_w).

		WTP_{air_long}	WTP_{fuel}	WTP_{eua}
WTP_{air_short}	*mean diff.*	196	235	415
	t-statistic	14.03^{***}	10.29^{***}	18.08^{***}
	effect size d	0.63	0.46	0.81
	effect size r_t	0.53	0.42	0.63
	z-statistic	14.46^{***}	11.18^{***}	16.79^{***}
	effect size r_w	0.46	0.35	0.53
WTP_{air_long}	*mean diff.*		39	220
	t-statistic		2.11^{*}	12.73^{***}
	effect size d		0.09	0.57
	effect size r_t		0.09	0.50
	z-statistic		4.40^{***}	14.38^{***}
	effect size r_w		0.14	0.46
WTP_{fuel}	*mean diff.*			181
	t-statistic			10.01^{***}
	effect size d			0.45
	effect size r_t			0.41
	z-statistic			9.93^{***}
	effect size r_w			0.31

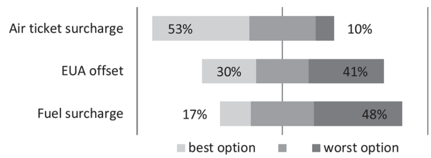

Fig. 5. Respondents' ranking of policy alternatives. The figure shows the response to the question: "Please rank the different strategies to price and reduce personal carbon emission" (n = 500). To compare, 75% of the respondents had a positive WTP for the air ticket surcharge, 50% for the fuel surcharge, and only 29% for EUA offsetting.

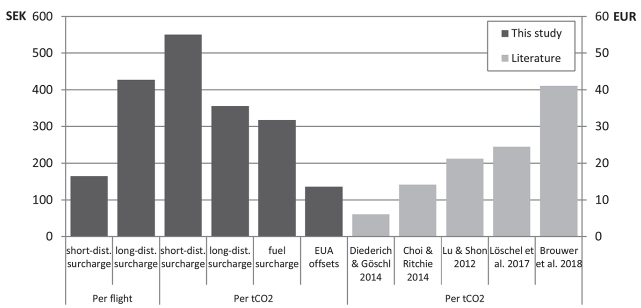

Fig. 4. Mean WTP for climate change mitigation in comparison to literature. WTP is shown per ton of CO_2 and per flight. Values from literature are shown in light grey.

Table 5

Robustness of mean differences if assumed carbon intensities are varied. All values in SEK/ tCO_2.***, ** and * indicate significance at 1%, 5% and 10%, respectively. Where the significance levels of *t*-test and Wilcoxon signed-rank test results differed, the significance of the latter was added in brackets. Results susceptible to changes in assumptions are highlighted in grey.

		Mean WTP	WTP_{air_short} 551	WTP_{air_long} 355	WTP_{fuel} 317	WTP_{eua} 136
WTP_{air_short}	−50%	1103	–	747***	786***	967***
	+/−0%	551	–	196***	235***	415***
	50%	368	–	12(**)	51***	232***
WTP_{air_long}	−50%	711	160***	–	394***	575***
	+/−0%	355	−196***	–	39*(***)	220***
	50%	237	−314***	–	−80***(**)	101***
WTP_{fuel}	−10%	352	−199***	−3(*)	–	216***
	+/−0%	317	−235***	−39*(***)	–	181***
	10%	288	−263***	−67***	–	152***

collection[4] as the governing coalition planned to introduce such a tax, which was in turn contested by various stakeholders. Against this background, some respondents were likely driven to think that the survey was 'not so hypothetical' and their answers could in fact influence policy choice. Hence, they might have understated their actual WTP or stated not to be willing to pay in principle in order to avoid costly policies in the future. It is impossible to quantify the strategic bias in this study, but both the focus in the survey design on mitigating hypothetical bias and the salience of carbon pricing (in particular air ticket taxation) among the Swedish public mean that there likely have been cases of strategic undervaluation. Several voluntary comments at the end of the online survey hint in the same direction (e.g. "hideous with all the extra taxes" or "the taxes we already pay for gasoline and fuel are already too high").

3.5. Robustness tests

The robustness of results was tested with respect to the assumed carbon intensities used in the conversion from WTP/ flight and WTP/ litre of fuel to WTP/ tCO_2. Varying these carbon intensities has a substantial impact on the mean WTP/ tCO_2 *levels* (see Table 5). With increasing carbon intensity, measured WTP goes down. Mean *differences*, on the other hand, remain overall significant. There are a couple of exceptions. When assuming a low carbon intensity of long-distance flying, mean WTP_{air_long} is significantly higher than mean WTP_{fuel}, but when assuming a high carbon intensity, this relationship is reversed ($WTP_{fuel} > WTP_{air\_long}$). Moreover, when halving the assumed carbon intensity of long-distance flying to 67 gCO_2/ pkm while keeping the carbon intensity of short-distance flying stable, WTP_{air_long} becomes significantly higher than WTP_{air_short}. It is, however, important to note that varying the carbon intensity for long-distance flights without changing the one for short-distance flights is a rather unlikely scenario.

Summing up, the robustness tests show that mean WTP for a surcharge on air tickets is, indeed, not very robust to large variation of carbon intensity, which in turn is caused by the scientific uncertainty about the GWP of emissions from aviation [48,67]. On the other hand, the two major differences between PVs, i.e. between voluntary and mandatory PVs and between PVs implying low costs and high costs, are robust to changes in the assumed carbon intensities.

[4] While there is no formal media study about this, a simple 'google trend' analysis (https://goo.gl/Me5Leh) showed that interest in the term 'flygskatt' (air ticket tax) spiked in Sweden in December 2016, not long before the online survey was started. Over the period November 2016 till Jan 2017 searches for 'flygskatt' were on average nearly as common as searches for 'flyktingkrisen' (refugee crisis), another important policy issue in Sweden.

4. Policy implications and conclusions

In the policy realm, CO_2 emissions are frequently treated as a negative externality of economic activity, a specific market failure that needs to be internalized. While there are challenges in determining the value of this externality, the so called social cost of carbon [68], carbon pricing levels consistent with the Paris Climate Agreement have been estimated to be 15–360 USD/ tCO_2-eq in 2030, 45–1000 in 2050, and 140–8300 in 2100 [69]. In theory, a uniform carbon price is the most cost-effective way to internalise the climate change externality [70,71]. However, current carbon prices are well-below required price levels [42], and a uniform explicit carbon price alone is insufficient in driving mitigation pathways consistent with the Paris Climate Agreement [69]. Also when looking back, the use of policies focussing on market failures and carbon pricing has not yet resulted in sufficient climate change mitigation [72,73].

The results of this study suggest that people do not value climate change mitigation with one uniform monetary value per each ton of CO_2, which is questioning the suitability (let alone political acceptability) of a uniform carbon price in a world that needs rapid and deep decarbonisation. The contextual and behavioural factors driving the variation in people's valuation of climate change mitigation have important implications for both the focus and the alternative design of policy interventions, and these implications are elaborated below.

First, this study provides orientation for policy (reform) priorities. In this case, the efforts to introduce an air ticket tax in Sweden were clearly supported: 75% of respondents were willing to pay an air ticket surcharge in principle, which is in line with a recent poll that found 73% approval for the planned air ticket tax [74]. Moreover, mean WTP_{air_short} was at 551 SEK/ tCO_2 substantial, even though still well below the current Swedish carbon tax of 1 150 SEK/ tCO_2 (which is not charged on aviation fuels). In contrast to the air ticket surcharge, only 50% of the respondents were willing to pay an additional surcharge on motor vehicle fuels and this surcharge was preferred least among the three PVs, indicating that this is not an area in which policy makers should expect very high public acceptance.

Second, as WTP appears to be higher for low-cost activities, differentiated carbon pricing can be justified, e.g. different surcharges for long-, medium- and short-distance flights. The planned level of the Swedish air ticket tax of 60 SEK per domestic flight appears, however, low compared to the mean WTP_{air_short} of 165 SEK per flight. The planned tax level for long-distance flights (400 SEK) is, on the other hand, closer to the mean WTP_{air_long} of 427 SEK per flight. It is important to note that if the (implicit) carbon price of a policy is low compared to the respective WTP, this may imply higher public acceptability but also less (expected) behavioural change, as people are still willing to pay the extra cost. This seems to be particular problematic for short-distance flights, which in contrast to long-distance travel cannot be considered a transportation 'need' in a climate-constrained world as transport alternatives exist [75]. Hence a strong price incentive could incentivise a shift towards these other modes of transport. Moreover, the effect of a weak carbon price incentive can be further decreased, as once there is a price to pay, people might feel less guilty about flying frequently since they ‘pay for it’. This is sometimes referred to as the ‘crowding-out’ of intrinsically motivated behaviour [76].

Third, and closely related, as WTP at least partly depends on the absolute level of associated costs, more people might be willing to pay if the initial carbon price is low (and later gradually increased over time). The experience with fuel taxation shows that, once introduced and established, raising fuel taxes further faces less resistance than establishing a significant tax to start with [77].

Fourth, the finding that mandatory PVs are associated with a higher WTP than voluntary PVs suggests that stressing the mandatory nature and collective effort of a carbon pricing policy may raise acceptability among individuals. It has been shown that people dislike the free-riding that might occur when voluntarily contributing to climate change mitigation [26,78]. So, it appears to be a good policy-promotion strategy to communicate clearly that a mandatory climate surcharge has to be paid by everyone and that it makes the worst polluters pay most. Even more importantly, the results support the finding that counting on voluntary offsetting does not appear to be a sufficient climate change mitigation strategy [79].

While above mentioned policy implications make a small contribution to evidence-based carbon pricing policy design, further research is needed on several questions. To start with, a better understanding of public acceptability of additional implicit carbon pricing interventions (including product standards and other regulations) is needed. Moreover, further insight about socio-psychological factors driving WTP in the context of implicit carbon pricing mechanisms can help to implement effective mitigation policies. Existing models for energy savings and human behaviour could be used as research (and intervention) framework [80]. Besides designing and implementing effective carbon pricing policies, communicating them in a good way to the public is an important aspect that has not received sufficient attention, yet [81]. The communication of carbon pricing policies includes subjects such as the framing of environmental impacts as losses or gains [82]. Finally, more knowledge about the magnitude of crowding out pro-environmental behaviour with carbon pricing mechanisms, in particular those with low price levels, is needed in order to obtain the intended mitigation effects.

Finally, and to conclude this paper, the findings of this study suggest some viable elements of a climate change mitigation policy mix: 1) charge carbon emissions where it is not done yet and where people are still willing to pay (e.g. for air travel); 2) these charges should be mandatory for all; 3) use revenues generated by additional carbon pricing to support the development of low-carbon substitutes in areas of energy consumption, in which WTP for climate mitigation is low. These three points are by no means sufficient, as modest explicit carbon pricing is not the panacea and further stringent regulations and mandates addressing implicit carbon pricing mechanism are needed for an effective policy mix (e.g. direct regulation, subsidies for low-carbon technologies, product standards). The points provide, however, a more behaviourally-informed starting point in the area of carbon pricing, acknowledging that there never is the will or capacity to implement a comprehensive and effective climate change mitigation policy mix all at once.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.erss.2019.01.022.

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Designing air ticket taxes for climate change mitigation: insights from a Swedish valuation study

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ABSTRACT

Research on air travellers' willingness to pay (WTP) for climate change mitigation has focussed on voluntary emissions offsetting so far. This approach overlooks policy relevant knowledge as it does not consider that people may value public goods higher if they are certain that others also contribute. To account for potential differences, this study investigates Swedish adults' WTP for a mandatory air ticket surcharge both for short- and long-distance flights. Additionally, policy relevant factors influencing WTP for air travel emissions reductions were investigated. The results suggest that mean WTP is higher in the low-cost setting associated with short-distance flights (495 SEK/ tCO_2; 50 EUR/ tCO_2) than for long-distance flights (295 SEK/ tCO_2; 30 EUR/t CO_2). The respondents were more likely to be willing to pay the air ticket tax if they were not frequent flyers, if they were women, had a left political view, if they had a sense of responsibility for their emissions and if they preferred earmarking revenues from the tax for climate change mitigation and sustainable transport projects.

Key policy insights

- A mandatory air ticket tax is a viable policy option that might receive majority support among the population.
- While a carbon-based air ticket tax promises to be an effective tool to generate revenues, its potential steering effect appears to be lower for low cost contexts (short-distance flights) than for high cost contexts (long-distance flights).
- Policy consistency regarding the tax base and its revenue use may increase public acceptability of (higher) air ticket taxes. Earmarking revenues is clearly preferred to tax recycling or general budget use.
- Insights about the personal drivers behind WTP for emissions reductions from air travel can help to inform targeting and segmentation of policy interventions.

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1. Introduction

In the current policy debate, carbon pricing mechanisms are a central strategy to steer consumption and investments towards low-carbon technologies and more sustainable practices (Baranzini et al., 2017; World Bank, Ecofys, & Vivid Economics, 2017). One sector where this approach is not very far developed, neither in terms of emissions covered nor in terms of the carbon price level, is aviation (Penner, 1999; Sims et al., 2014). To date, only a few countries (e.g. UK and Norway) have introduced direct taxes or charges on air travel, and the aviation sector even benefits from significant subsidies (Gössling, Fichert, & Forsyth, 2017). In the EU, intra-continental flights are covered by the EU emissions trading system (ETS). However, flights into and out of the EU are not included in the ETS, over 80% of ETS emissions allowances (EUA) are allocated for free to the aviation sector, and carbon prices under the EU ETS have been low, which means that the inclusion of

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air travel in the ETS has not yet resulted in a significant carbon price signal for air travel (Cui, Li, & Wei, 2017; Meleo, Nava, & Pozzi, 2016). Internationally, the 'Carbon Offsetting and Reduction Scheme for International Aviation' (CORSIA) that was initiated by the International Civil Aviation Organization (ICAO) aims for carbon neutral growth of the sector after 2020. However, the scheme will only cover additional emissions above the 2020 level, it will be a voluntary scheme to start with, and it will not apply to domestic flights (ICAO, 2016).

Driven by the lack of stringent pricing policies, a common way to address greenhouse gas emissions from air travel is to encourage voluntary carbon offsetting (Daley & Preston, 2009). However, considering the need to cut aviation emissions by half in order to stay within 2°C of warming by mid-century above pre-industrial levels, voluntary offsetting is an insufficient approach (Becken & Mackey, 2017). Emissions from aviation are projected to grow by 140% between 2013 and 2050 (Kuramochi et al., 2018), when the contribution of aviation to global CO_2 emissions may reach 22% (Cames, Graichen, Siemons, & Cook, 2015). The current share of emission offsets in total emissions from air travel is negligible (Zelljadt, 2016), and the additionality and mitigation potential of offsetting have been questioned (Broderick, 2008). Another policy instrument is to tax air travel, thereby providing an incentive to travel less (Daley & Preston, 2009; Sims et al., 2014). Today, the most common way to create this incentive is air ticket taxation (Krenek & Schratzenstaller, 2016) since charging VAT on international flights or taxing kerosene would first require (re-) negotiations of international agreements.

A complication in the design of air ticket taxation is that there is little information about people's willingness to pay (WTP) for such mandatory taxes. In contrast, various studies have investigated air travellers' WTP for *voluntary offsets* (Brouwer, Brander, & Van Beukering, 2008; Choi & Ritchie, 2014; Jou & Chen, 2015; Lu & Shon, 2012; MacKerron, Egerton, Gaskell, Parpia, & Mourato, 2009). However, this information is not particularly useful for the design of air ticket taxes as there are strong indications that WTP is systematically lower for voluntary offsets than for coercive instruments (Segerstedt & Grote, 2016; Wiser, 2007).

By addressing this shortcoming, this study is the first that investigates WTP for mitigating air travel emissions based on a mandatory payment vehicle, a climate surcharge on air tickets. In doing so, the study aims to improve the valuation of mitigating air travel emissions and to increase its policy relevance. The study further adds to the existing body of work on payment vehicles by contrasting WTP in low cost contexts (short-distance flights), to that of high cost contexts (long-distance flights). Moreover, it investigates policy relevant aspects driving people's WTP besides socio-demographic factors, including political views, flight frequency, sense of responsibility for emissions, as well as preference for earmarking revenues from carbon pricing policies. Knowing the factors that drive WTP for mitigating air travel emissions in a specific context can support the design of effective policy interventions and increase their public approval.

The specific context of this study is Sweden and its policies to address the climate impact of air travel. At the time of the study (early 2017), the introduction of a climate tax on air tickets was publicly debated in Sweden. The ticket tax was eventually introduced in April 2018. In the research informing the preceding debate and the legislative process (Andersson & Falck, 2017), data about the WTP of Swedes for mitigating emissions from flying was limited to people's general WTP for offsets (Gössling, Haglund, Kallgren, Revahl, & Hultman, 2009). Moreover, Swedes' WTP for climate mitigation in general had been estimated (Carlsson et al., 2012), but there was no evidence regarding the specific WTP for air travel emissions. Thus, with respect to Swede's WTP for mitigating air travel emissions, the existing literature was and still is fragmented and does not provide conclusive answers to at least three questions: What is (approximately) Swedes' WTP for the mitigation of their air travel emissions in a mandatory scheme? Is there a difference in WTP elicited between short distance flights and long distance flights? And what are the factors influencing WTP? In order to respond to these questions, empirical data was collected in a contingent valuation survey, which is presented in the following section, together with the approach for data analysis. This is followed by Section 3 which presents and discusses the results of the survey and econometric estimations. Section 4 presents policy implications and concludes.

2. Data and methods

2.1. Contingent valuation (CV) survey

The data for this study is based on a contingent valuation survey ($n = 500$), which was implemented by computer-assisted web interviewing of a representative, random sample of Swedish adults in January 2017.[1] All respondents went through the WTP elicitation process repeatedly in order to elicit WTP for emissions reductions with different payment vehicles (PVs), including a mandatory air ticket surcharge on short-distance flights and the same surcharge on long-distance flights.[2] The order of all PVs in the survey was randomized in order to avoid bias induced by order or fatigue effects. To elicit WTP, the survey combined a simple dichotomous choice question about general WTP with an iterative bidding process for those respondents that were willing to pay in principle. Iterative bidding followed a pre-defined bidding structure (see Figure 1).

The level of bids was the same for all PVs and ranged from SEK 100–1,000[3] per ton CO_2, which reflects the results of previous CV studies (Brouwer et al., 2008; Löschel, Sturm, & Uehleke, 2017) and the price range of current carbon pricing schemes (World Bank, Ecofys, & Vivid Economics, 2016). Bid values per tCO_2 were transformed to corresponding values per short-distance flight (30–300 SEK) and long-distance flight (120–1,200 SEK). This transformation assumed a carbon intensity of 171 gCO_2/ pkm for the short-distance (ca. 1,750 km) example flights and 133 gCO_2/ pkm for the long-distance (ca. 9,000 km) example flights. These carbon intensities are high compared to industry estimates (Andersen Resare, 2015), in order to partly account for greenhouse gas emissions other than CO_2 that are emitted from flying (see Section C of the supplementary material for further elaboration on these carbon intensities). Because of the uncertainty connected to carbon intensity estimates, the sensitivity of WTP results to changes in the assumed carbon intensity was tested. In the WTP elicitation, no reference was made to the price of the air tickets (see text in Figure 2), since ticket prices are not fully proportionate to travel distance, and hence price-anchors would have potentially biased the results.

The interpretation of WTP responses followed a conservative approach. Those who were not willing to pay in principle and those who rejected the lowest bid were considered to have a WTP of zero. For all others, the highest accepted bid was treated as their WTP. Respondents could also state that they are uncertain about a bid, which was treated as rejection in order to reduce potential hypothetical bias (Loomis, 2014). Respondents who indicated that their WTP was higher than the highest bid of 300 and 1,200 SEK for short- and long-distance flights respectively could freely state their maximum WTP up to a cap of four times the highest bid. This cap was chosen in order to avoid unrealistically high valuations and still make sure that most people's values are covered, even for rather extreme results (e.g. mean WTP of 900 SEK/t and a standard deviation of >1,000 SEK/t).

Besides uncertainty recoding and capping the highest bid, further measures were taken to counteract hypothetical bias. These measures included consequentiality design (Loomis, 2014) of the survey ('results of this survey are meant to inform policy reforms'), and a reminder about the opportunity cost of paying for CO_2 emissions ('don't agree to costly policies if you think you cannot afford it or if you feel that there are more important things for you to spend your money on'). Strategic bias, on the other hand, might entail that respondents systematically undervalue a good in order to avoid costly policies in the future (Venkatachalam, 2004). This potential bias was addressed by reminding respondents that they are asked about their personal

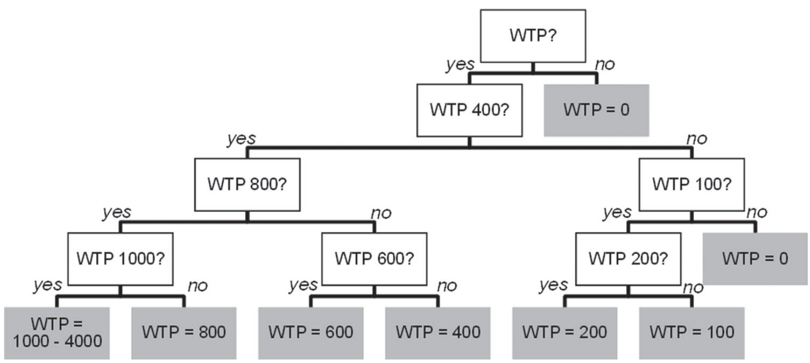

Figure 1. Pre-defined bidding structure. The numbers indicate bid values in SEK/tCO_2. Grey boxes show the endpoints, i.e. the WTP values that were used for further analysis.

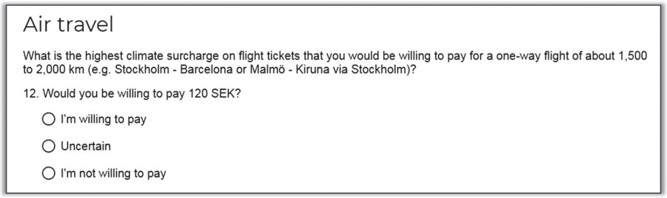

Figure 2. Example for WTP elicitation question. This is translated from the original survey which was conducted in Swedish.

WTP and not what they think is the right level for pricing air travel emissions in general (section D of the supplementary material contains more information about how different biases were addressed in the survey design).

Besides the elicitation of WTP, the survey also included questions about socio-demographic data, travel behaviour and policy preferences in the context of carbon pricing. The survey data is summarized in Table 1, which also includes the variable codes, means and standard deviations.

2.2. Econometric analysis

An econometric analysis was carried out to identify significant drivers of respondents' general and specific WTP for a surcharge for their air travel emissions. General WTP refers to respondents' dichotomous choice to support the surcharge or not ($WTP_{surcharge}$), while specific WTP refers to the amount respondents are willing to pay for the surcharge on a short- and long-distance flight ($WTP_{airshort}$, $WTP_{airlong}$). First, a simple logit regression approach

Table 1. Coding, mean and standard deviation (SD) of variables.

Variable *(name)*	Codes and explanation	Mean	SD	Population mean
Age *(age)*	25–74 (number in years)	50.54	14.11	48.48
Gender *(female)*	0 (male or other), 1 (female)	0.49	0.50	0.49
Household size *(hhsize)*	1 (1 person lives in household) to 6 (more than 5 people)	2.45	1.22	2.2*
Monthly income before tax *(income)*	1 (< 10 000 SEK), 2 (10 000–19 999 SEK) to 8 (> 70 000 SEK)	3.45	1.34	**
Education *(education)*	1 (elementary school) to 5 (licentiate or PhD)	3.05	1.05	
Left political view *(leftpolview)*	1 (clearly to the left or left), 0 (right-leaning, neither left nor right, not shared)	0.34	0.47	0.41***
Frequent flier *(frequentfly)*	1 (fly several times per year), 0 (fly 1 or less times per year)	0.30	0.46	
Responsible *(responsible)*	1 (ranked air travellers 1st or 2nd among 5 actors potentially responsible for reducing emissions), 0 (ranked 3rd, 4th or 5th)	0.25	0.43	
Earmark *(earmark)*	1 (preferred revenue use is for climate mitigation or sustainable transport), 0 (preferred use is for general budget or revenue recycling)	0.77	0.42	
Dependent variables				
General WTP for surcharge *($WTP_{surcharge}$)*	1 (positive WTP in principle), 0 (no WTP)	0.75	0.43	
WTP short-distance air surcharge *(WTP_{air_short})*	0 ($WTP_{surcharge}$ = 0 or lowest bid was rejected), 100–4000 (the respective WTP value in SEK/t CO_2)	495	519	
WTP long-distance air surcharge *(WTP_{air_long})*	0 ($WTP_{surcharge}$ = 0 or lowest bid was rejected), 100–4000 (the respective WTP value in SEK/t CO_2)	295	374	

*mean for the whole Swedish population (not only the ones aged 25–74).
**median income before tax was 28 500 SEK for respondents and 29 100 SEK for the general population (assuming a tax rate of 30%; for the age group 25–64).
***in the 2018 national elections the left block of parties received 40.7% of the votes.

was chosen to identify the significant predictors for $WTP_{surcharge}$. Second, interval regressions were carried out for $WTP_{airshort}$ and $WTP_{airlong}$. Interval regression accounts for the fact that the data for $WTP_{airshort}$ and $WTP_{airlong}$ consists of intervals (between highest accepted and lowest rejected bid); and it is has been previously applied in a comparable context (Brouwer et al., 2008). Finally, an interval regression model was computed that combined WTP responses for short- and long-distance flights by using the lower of the two values[4] as lower bound and the higher value as upper bound of respondents' intervals ($WTP_{aircombined}$). It is important to note that the WTP data were transformed to SEK/ tCO_2 values in order to make the coefficients comparable across the three interval regressions.

The model specifications for these regressions were based on a literature review that was carried out to identify policy relevant factors affecting people's views and preferences towards climate change mitigation. The review resulted in a list of variables including respondents' air travel frequency (Brouwer et al., 2008), *frequentfly*; their political view (Hornsey, Harris, Bain, & Fielding, 2016), *leftpolview*; their sense of responsibility for their emissions (Brouwer et al., 2008), *responsible*; and their preference for earmarking revenues from carbon pricing (Kotchen, Turk, & Leiserowitz, 2017), *earmark*. For all of these variables, data was collected in the survey (see Table 1 above).

In order to explore their potential explanatory power, partial correlation tests were carried out, controlling for a set of socio-demographic variables (*age*, *female*, *education*, *hhsize*, *income*). The tests showed highly significant partial correlations to $WTP_{surcharge}$, $WTP_{airshort}$ and $WTP_{airlong}$ for all four variables (*frequentfly*, *leftpolview*, *responsible* and *earmark*). Therefore, all four were included in the following conceptual model for regression analysis:

$$y = X_1\beta_1 + \ldots + X_9\beta_9 + \varepsilon \tag{1}$$

where y is $WTP_{surcharge}$, X_1 to X_5 are above mentioned socio-demographic characteristics, X_6 is flight frequency (*frequentfly*), X_7 is the political view (*leftpolview*), X_8 is the sense of responsibility for one's emissions (*responsible*) and X_9 is the preference for earmarking of tax revenue (*earmark*); β_1 to β_9 are the corresponding coefficients, and ε is an error term. After estimating the full model (1) also a stepwise removal of variables that were below the 10% significance level ($p < 0.1$) was executed, both for the logit regression of $WTP_{surcharge}$ and for the interval regressions of $WTP_{airshort}$, $WTP_{airlong}$ and $WTP_{aircombined}$.

3. Results and discussion

Below, first the results from WTP elicitation are presented and discussed, including differences between $WTP_{airshort}$ and $WTP_{airlong}$, and the sensitivity of results to changes in carbon intensity of flying. Second, the econometric analysis is presented and results are discussed. Finally, contextual factors from the specific situation in Sweden are included in the discussion.

3.1. Willingness to pay for emissions from air travel

The survey shows that more than 70% of respondents had a positive WTP for their air travel emissions (see Figure 3). The distribution of positive WTP shows that the approval was highest at the first (and central) bid of the iterative bidding process (at 120/ 480 SEK per short-/ long-distance flight), indicating a potential anchoring effect. In contrast, the distribution of WTP responses above and below the central value sharply differs between short- and long-distance flights. While for short-distance flights only 13% of respondents indicated a positive WTP below the central bid, this share was 29% for long-distance flights. WTP above the central bid was, in turn, much more frequent for short-distance flights (42%) than for long-distance flights (17%).

This difference between short- and long-distance flights is also reflected in average WTP values. After transforming WTP values in order to receive comparable results in SEK/ tCO_2, average WTP for long-distance flights turned out to be much lower than for short-distance flights (295 versus 495 SEK/ tCO_2). It was shown in a two-sided t-test ($t = 13.7$, $p < 0.001$) and a Wilcoxon signed-rank test ($z = 14.5$; $p < 0.001$) that the difference between the two means of SEK 200 is statistically highly significant. This finding is in line with previous research which suggests that WTP is higher in a low-cost decision context (Blasch & Farsi, 2014; Diekmann & Preisendörfer, 2003). While bid levels *per* tCO_2 were the same for both short- and long-distance flights, bids *per flight* were

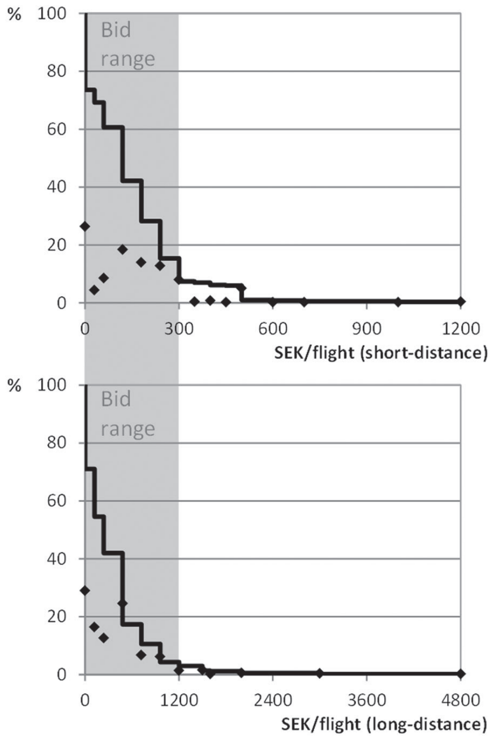

Figure 3. WTP distribution for short- and long-distance flights. Diamonds indicate the respective shares of acceptance at given bid levels; solid lines indicate cumulative acceptance. The grey area marks the survey's bid range. For both graphs X-axis values correspond to a range of 0–4 000 SEK/ tCO_2.

four times lower for short-distance flights. This influence of the absolute cost-context on WTP illustrates the importance of eliciting WTP for the same good in different cost-contexts, or making at least the cost-context explicit when reporting results.

Average WTP_{air_short} and WTP_{air_long} per tCO_2 are at 495 and 295 SEK (ca. 50 and 30 EUR) still within the range found in previous studies, including 41 EUR for European air travellers (Brouwer et al., 2008), 14 EUR (21 AUD) for Australian air travellers (Choi & Ritchie, 2014), and 21 EUR (25 USD) in the Taiwanese context (Lu & Shon, 2012). The WTP values presented in Figure 4 are, however, conservative estimates, as respondents' highest accepted bid was used as their maximum WTP. If, instead, the midpoints of the intervals between highest accepted and lowest rejected bid are used, WTP_{air_short} and WTP_{air_long} increase to 551 and 353 SEK respectively. One explanation for the comparably high WTP estimates found in this study is that it used a mandatory PV while previous studies were based on voluntary offsetting. This is in line with previous research in the context of climate change mitigation, which found that mandatory PVs are associated with higher acceptance (Segerstedt & Grote, 2016) and higher WTP (Wiser, 2007) than voluntary PVs.

Absolute WTP results of this study (and previous research) should be treated with due scepticism. This is not only due to the effect that framing of PVs may have, but also due to the very specific case settings of CV studies and their (often) limited sample sizes. Moreover, WTP levels for air travel emissions also depend on the assumed carbon intensity of air travel. While WTP in this study was elicited per flight (ticket), the transformation to WTP per tCO_2 requires a conversion factor. This carbon intensity factor, measured in gCO_2/pkm, depends among others on aircraft efficiency, capacity utilization, and assumptions about the global warming potential of air

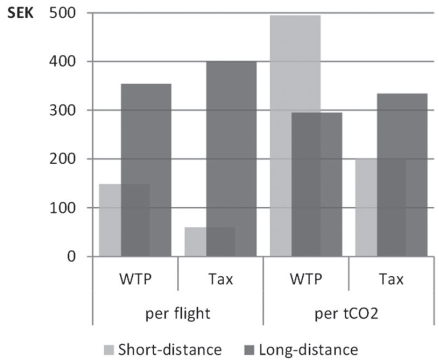

Figure 4. Juxtaposition of average WTP values and Swedish air ticket tax for short and long distance flights. The figures per ton CO_2 were achieved by dividing WTP per flight and tax levels (SEK 60 short-distance and SEK 400 long-distance) by the CO_2 emissions caused by the example-flights from the survey (0.3t short-distance and 1.2t long-distance).

travel. The sensitivity analysis for average WTP_{air_short} and WTP_{air_long} presented in Figure 5 shows that changes in assumed carbon intensity strongly impact average WTP levels. However, the difference between WTP_{air_short} and WTP_{air_long} found in this study remains significant for all plausible combinations of carbon intensity.[5] The finding that WTP is higher in a low-cost setting is, hence, robust to changes in (assumed) carbon intensity.

3.2. Personal factors influencing WTP

The regression analysis revealed that the variables *frequentfly*, *leftpolview*, *responsible* and *earmark* are all significant predictors of WTP, while from the socio-demographic variables only *female* is a significant predictor of $WTP_{surcharge}$ and *income* is a significant predictor of $WTP_{airshort}$, $WTP_{airlong}$ and $WTP_{aircombined}$ (see Table 2). The latter effect of *income* on WTP was found in many previous CV studies, also in the context of flying and emissions offsetting (Brouwer et al., 2008; Jou & Chen, 2015; Löschel et al., 2017). The variable *female* significantly increased the *likelihood* of being willing to pay ($WTP_{surcharge}$), which supports previous evidence for gender differences in environmental behaviour in general (Zelezny, Chua, & Aldrich, 2000) and in WTP for air travel emissions in particular (MacKerron et al., 2009).

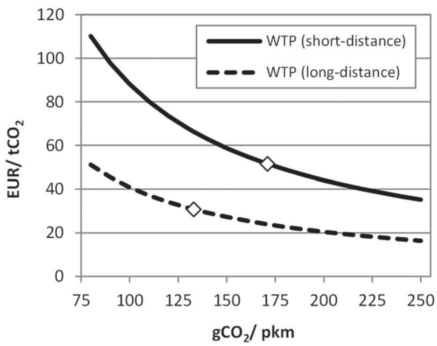

Figure 5. Sensitivity of mean WTP to changes in carbon intensity. Diamonds indicate the values used in this study. For comparison, the Swedish carrier SAS communicates average carbon intensity of 100 gCO_2/pkm (Andersen Resare, 2015), while CO_2 equivalent emissions are estimated to be higher by a factor of up to 2 (Lee et al., 2010).

Of the other significant variables, *frequentfly* is the only one with a negative sign. In the context of $WTP_{surcharge}$, this backs the recent finding that support for aviation policies is weaker among 'aeromobile' people (Kantenbacher, Hanna, Cohen, Miller, & Scarles, 2018). On the other hand, people who feel personally *responsible* for their own emissions tend to agree with the surcharge ($WTP_{surcharge}$). This is in line with the finding that a perception that the general public is responsible for climate change (in contrast to governments or industry) significantly increases the approval of mandatory carbon offsetting (Kantenbacher et al., 2018). Feeling *responsible* is also associated with an increase in the amount people were willing to pay, again consistent with previous research (Brouwer et al., 2008). It is important to note that only one quarter of respondents ranked themselves (i.e. air travellers) first or second when asked who from a group of five different actors is most responsible to reduce emissions from air travel. This closely replicates corresponding findings from a survey study at a Swedish airport (Gössling et al., 2009).

Besides the personal sense of responsibility, also people's political view appears to be significant for WTP. A *leftpolview* is associated with a higher likelihood for $WTP_{surcharge}$. This, however, only implied a modest political polarization on the issue. While $WTP_{surcharge}$ was, indeed, highest among people with a clearly left political view (82%), it was still above 50% across the whole political spectrum. A recent opinion poll found similar approval of the Swedish air ticket tax but a slightly more pronounced political polarization (Rosén & Kihlberg, 2018).

Finally, a preference for earmarking tax revenues for climate change mitigation or sustainable transport solutions (*earmark*) is positively associated with WTP. This finding is in line with previous research that earmarking is the preferred option to use revenues form carbon pricing policies (Baranzini & Carattini, 2017; Drews & Bergh, 2016; Kotchen et al., 2017).

3.3. Potentially influential factors from the Swedish context

In addition to personal factors influencing WTP, two key aspects of this study's Swedish context were identified that may have influenced the results: (a) the lively public debate around a climate tax on air travel, and (b) socio-cultural aspects of environmental policy making in Sweden.

First, the public debate in the context of the Swedish air ticket tax might have led to strategic behaviour among respondents. At the time of the study in early 2017 an air ticket tax had been publicly discussed and planned for by the Swedish government for several months. Therefore, the air ticket surcharge that was presented in the survey was likely a policy that was perceived by respondents as something that might actually be implemented, hence giving them an incentive to strategically undervalue emissions reductions under a

Table 2. Results from stepwise regression of model (1).

Variable	Logit	Interval regression		
	$WTP_{surcharge}$	$WTP_{airshort}$	$WTP_{airlong}$	$WTP_{aircombined}$
female	0.383*			
	(0.224)			
income		44.8**	34.0***	35.2***
		(17.5)	(12.5)	(13.4)
frequentfly	−0.655***	−103.5**		−66.2*
	(0.231)	(50.9)		(39.2)
leftpolview	0.894***	180.9***	148.6***	156.9***
	(0.265)	(48.0)	(35.7)	(37.1)
responsible	0.865***	157.1***	136.9***	142.1***
	(0.298)	(52.1)	(38.7)	(40.4)
earmark	1.145***	168.1***	135.2***	162.1***
	(0.240)	(54.0)	(40.1)	(41.4)
constant	−0.115	196.3**	45.6	115.8*
	(0.250)	(78.9)	(58.6)	(60.4)
Pseudo R^2	0.117			
$\chi^2(9)$	65.6***	45.5***	51.4***	58.3***
Log likelihood	−248.4	−2289.9	−1959.9	−2047.8

Note: The significance level for removal was $p < 0.1$. Standard errors in brackets. ***, ** and * indicate significance at 1%, 5% and 10%, respectively.

mandatory surcharge. Due to this strategic bias the level of WTP_{air_short} and WTP_{air_long} might in fact be higher than stated by respondents.

On the other hand, there are some sociocultural particularities of Sweden, which might have led to higher WTP (than could be expected in other countries). Case or country specific social and cultural aspects are highly relevant for the acceptance of climate change policies (Alló & Loureiro, 2014). Sweden is a country characterized by a high trust in government (Rothstein, 2015), a high environmental awareness among the population, particularly with respect to climate change (EC DG Communication, 2017), and a strong tradition of environmental taxation (OECD, 2014). This implies that a government-administered tax or surcharge targeting climate change is likely to be better accepted (also at a higher tax level) in Sweden than in countries with lower trust in government, lower environmental awareness and dislike of (environmental) taxation.

4. Conclusions and policy implications

The results of this study have implications both for carbon pricing of air travel in general and for the specific policy of air ticket taxation in Sweden. The first policy relevant finding is that *there is a considerable positive WTP for a mandatory surcharge on air travel emissions*. Compared to voluntary offsetting, for which previous studies also found positive WTP, a surcharge or tax is a favourable instrument from a climate mitigation perspective as it actually forces people to pay. In the case of offsetting there seems to be an attitude-behaviour gap (Higham, Reis, & Cohen, 2016; Juvan & Dolnicar, 2014), which might explain the discrepancy between stated WTP and the low actual participation in the offsetting market (Zelljadt, 2016). This study also provides some indications for the attitude-behaviour gap, as about half of the respondents expressed positive WTP, but did not feel that it is mainly the air travellers' responsibility to reduce emissions.

A tax that forces every air traveller to pay for the emissions has two potential climate impacts. First, it may steer behaviour away from flying. Second, it generates revenue that can be used for climate mitigation purposes. In both cases it is important that the incentive structure for different travel distances is well-designed. The new air ticket tax in Sweden is SEK 60 (EUR 6) for short-distance flights (including domestic and intra-European), SEK 250 (EUR 25) for medium-distance flights and SEK 400 (EUR 40) for long-distance flights (Swedish Tax Agency, 2018). If measured in SEK/t CO_2 (see Figure 4), the tax is higher for the long-distance flights used in this study (ca. 330 SEK/ tCO_2) than for the short-distance flights (ca. 200 SEK/ tCO_2). If the aim is to steer travel behaviour away from flying, and if distance-specific WTP values are taken into consideration, it should be the other way around and the *tax (per tCO_2) should be higher for short-distance flights than for long-distance flights*.

The air ticket tax is, however, only one policy with impact on the ticket price. In addition, the EU ETS applies to all intra-European (i.e. only short-distance) flights. The associated cost per flight has so far been negligible (<1 EUR per flight) as more than 80% of EUAs are allocated for free to the aviation sector and carbon prices under the EU ETS have been low (De Bruyckere & Abbasov, 2016). Moreover, also the reduced VAT rate of 6% is charged on Swedish domestic flights[6], but not on international flights. As international flights make up 80% of the passengers (and 90% of CO_2 emissions) of Swedish air travel (Kamb, Larsson, Nässén, & Åkerman, 2016), and as VAT is not relevant for business travel, the (reduced) VAT applies only to a small fraction of flights. So, even if the EU ETS, domestic VAT and air tickets taxes are all taken into consideration, the implicit carbon price on short-distance flights likely remains much lower than the implicit carbon price on long-distance flights. Considering further that environmentally preferable substitutes, such as high-speed trains, are only viable for short-distance travels, the incentive structure of the new air ticket tax seems to be misguided.

Moreover, the overall implied carbon price level on air travel is still low. With increasing EU ETS allowance prices and decreasing free allocation, the costs per short-distance flight may increase in the future, but the risk for 'over-charging' short-distance flights due to overlaps between the EU ETS and the air ticket tax is limited. Carbon prices implied by the air ticket tax (< 350 SEK) and on the market for EUAs (ca. 200 SEK in September 2018) are, for instance, still far away from the level of the Swedish carbon tax of 1,150 SEK/ tCO_2. The carbon tax is at the same time the most common value for the external cost of carbon used in Swedish transport planning (Trafikverket, 2016). Accepting the carbon tax as a valid proxy for the external cost of carbon in the Swedish context implies that *WTP values found in this study are lower bound estimates for the value of air travel emissions and that existing carbon pricing policies in the aviation sector are not ambitious enough*.

Accordingly, strong behavioural change is not expected and the Swedish tax is projected to reduce the number of flights only by about 3%, while annual growth until 2022 is projected to be 3.6% (Andersson & Falck, 2017). The main argument for keeping the Swedish tax at a relatively low level has been to avoid the risk of passengers shifting from domestic airports to airports abroad (Swedish Government, 2018), which was one of the reasons for abolishing the Dutch air passenger duty (Gordijn, 2010). The risk for demand shifts abroad is likely to be higher for long-distance flights, since air ticket taxes for these flights are higher. This is, in turn, an additional argument in favour of somewhat higher taxes on short-distance flights.

The low overall low tax level, the relatively low tax rate on short-distance flights and the risk of demand shifting to other countries suggest that the effectiveness of a tax scheme relies on the use of the revenues for climate mitigation, rather than on behavioural change. Yet, revenues from the new Swedish air ticket tax, projected to be SEK 1.8 billion per year, are programmed to go to the general budget (Andersson & Falck, 2017). Similarly, revenues from EU ETS auctions in Sweden are also not earmarked (Le Den, Beavor, Porteron, & Ilisescu, 2017). This not only reduces the effectiveness of the tax or the trading scheme, but general budget use was also the least popular option for revenue use among respondents of the survey. Respondents' preference for earmarking was significantly associated with their WTP, which implies a wish for consistency between the tax base and the intended revenue use. These findings suggest that *earmarking revenues may raise public acceptance or might enable a more ambitious pricing policy*. A recent study even suggests that there is an association between the expected (direct) environmental effectiveness of a tax and the preference for environmental earmarking (Carattini, Baranzini, Thalmann, Varone, & Vöhringer, 2017). Hence, the strong preference for earmarking found in the Swedish case might well be explained by respondents' low expectations for the effectiveness of the air ticket tax.

In addition to increased effectiveness, earmarking revenues would address another common criticism of the Swedish air ticket tax, namely its lack of incentives for innovation. The air ticket tax has been criticized on the grounds that, by taxing tickets rather than emissions or fuels, it does not encourage innovation. Economic modelling has shown that air ticket and fuel taxes both impact emissions, but that only fuel taxes incentivize fuel economy improvements, and that there are no clear-cut differences in welfare effects between the two (Keen, Parry, & Strand, 2013). The main reason for not taxing emissions, e.g. through fuel taxation, is related to international trade regimes limiting fuel taxation in aviation (e.g. European Council Directive 2003/96/EC). Earmarking of revenues may partly compensate for this by enabling the provision of funds for innovations related to, for instance, efficiency improvements and alternative fuels.

In addition to the effectiveness of air ticket taxation and its innovation potential, fairness is another important policy aspect. This study found that frequent fliers tend to have a lower WTP, which is problematic as it might be driven by free-riding, i.e. strategically understating WTP to avoid costly payments in the future (Venkatachalam, 2004). Frequent flyers are not only willing to pay less, but they also disproportionately use the highly subsidized aviation system (Gössling et al., 2017) and cause a larger amount of CO_2 emissions, thereby adding further costs to society. The mismatch between frequent flyers' lower WTP and higher impact imply that *it is a large challenge to implement adequate carbon pricing policies for frequent flyers*. This is a political rather than a technical challenge. There are proposals to, for instance, differentiate taxes between economy, business and first class tickets, or to introduce progressive tax rates that increase with flight frequency (Chancel & Piketty, 2015; Krenek & Schratzenstaller, 2016).

In conclusion, neither voluntary emissions offsetting, nor the inclusion of aviation in emissions trading schemes, nor the current practice of revenue use appear to be sufficient to counterbalance emissions growth from air travel. In contrast, air ticket taxes whose revenues are used for climate change mitigation appear to be a viable policy option as long as there is no ambitious international scheme in place. However, the substantial positive WTP for such a climate surcharge on air tickets also indicates that modest taxation will unlikely steer behaviour away from flying, and high tax rates and earmarking of revenues are needed in order to achieve considerable CO_2 emissions reductions.

Notes

1. The whole survey is available in Section A. of the supplementary material online; and Section B. of the supplement contains further information about the sampling process.

2. Two additional PVs, that were part of the survey but are not reported here, were a motor vehicle fuel surcharge and voluntary offsetting of transport emissions by purchasing and retiring EU emission allowances.
3. 10 SEK were about 1 EUR at the time of the study.
4. These two values were the respective midpoints of respondents' WTP intervals for short and long distance flights.
5. The analysis did not go beyond combinations in which the carbon intensity of short distance flights was double the one of long distance flights (e.g. 75/150, 100/200 and 125/250 g/pkm respectively).
6. Note that charging reduced VAT can be regarded a subsidy, rather than a tax.

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Green growth policies dominate the climate change mitigation discourse. But how much can they contribute to rapid decarbonisation? How viable is their economic growth objective? And can new behavioural insights be used to improve the effectiveness of green growth climate policies such as carbon pricing? These questions are addressed in this thesis by conducting economic policy analysis that goes beyond short-term cost-effectiveness and takes into consideration the actual track-record of green growth policies and a behaviourally realistic model of decision-making. The results of this thesis imply that green growth climate mitigation policies are not sufficient for reaching the ambitious targets of the Paris Agreement.

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