An investigation into the compatibility and effectiveness of linking Brazilian carbon mitigation and trading strategies with the EU ETS (European Union Emissions Trading Scheme)

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Doctor of Philosophy

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Dr. John Bartlett.

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To my baby angel, who brought me strength and will always be remembered with love. And to Ricardo, whose love and support encouraged me throughout this period in Ireland.

DECLARATION

This thesis has been developed while registered at the Institute of Technology Sligo for the degree of Doctor of Philosophy. It is entirely my work, which has not been submitted to any other degree at this or any other institution. Portions of this research have been submitted as four journal papers and four conference papers. These are listed below.

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LIST OF PUBLICATIONS

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Conference papers

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OLIVEIRA, T.D; GURGEL, A.C; TONRY, S. The effects for Brazil of Linking Emissions Trading Schemes in the context of the Heterogeneity of Trading Partners. In: 21st Annual Conference on Global Economic Analysis, 2018, Cartagena, Colombia. Conference Proceedings, 2018.

OLIVEIRA, T.D; GURGEL, A.C; TONRY, S. A Linked Emissions Trading Scheme under alternative scenarios: implications for Europe and Brazil. In: 3rd

Conference of the Hellenic Association of Energy Economics, 2018, Athens, Greece. Conference Proceedings, 2018.

OLIVEIRA, T.D; GURGEL, A.C; TONRY, S. Linking emissions trading schemes: an economic impact assessment for Europe and Brazil using EPPA6 model. In: 2017 Conference of the Ecomod network, 2017, Ljubljana, Slovenia.

“When you want something, all the universe conspires in helping you to achieve it”

PAULO COELHO, THE ALCHEMIST

“Aprendi que a coragem não é a ausência de medo, mas o triunfo sobre isso”

NELSON MANDELA

ABSTRACT

International climate negotiations have recently underlined the relevance of international cooperation via carbon pricing, as well as the need for support from developed and developing countries to tackle climate change. With Emissions Trading Schemes (ETS) emerging in developed and developing regions around the world, linking these systems may become a future option. This raises the question as to the appropriateness of such an approach. Based on discussions regarding the use of market-based instruments in Brazil, this thesis investigates the impact of a hypothetical ETS covering electricity and energy-intensive sectors in Brazil, using a global economy-wide model - the EPPA6. Land Use, Land-Use Change and Forestry (LULUCF) related emissions, which are significant in Brazil, are excluded from trading in an effort to closely align with existing provisions of the European Union Emissions Trading Scheme (EU ETS). We simulate linkages for Brazil with a developed region (Europe) and two developing regions (Latin America and China) under different ETS design scenarios. There are substantial differences in relation to the volume of emissions, the emissions profile and abatement opportunities between jurisdictions. Results of simulations for the 2020-2050 horizon show that the level of ambition is a key determinant on the effects of the ETS, as is trading partner compatibility. If Brazil is committed to an ambitious mitigation target, the strategy to link with other ETS systems is recommended. If linked to a developed country ETS, an inflow of revenues is envisaged from selling allowances in the long term, and it is possible to curb emissions whilst changing energy use patterns towards less carbon-intensive technologies. On the other hand, due to the contrasting stringency of targets, linking emissions mitigation and trading strategies with developing countries is economically more efficient, as it reduces the adverse impacts on the Brazilian economy. In this case, Brazil presents an import-oriented profile for emissions permits across the period. For trading partners, the benefits of linking with Brazil are mostly political, although there are associated gains from trading. Accordingly, there are advantages and disadvantages associated with each proposed trading situation. Within a linkage agreement, a more cost-effective ETS design to seize mitigation opportunities for Brazil includes a less stringent cap, or the introduction of revenue recycling for the production of alternative energy. Finally, the ETS should consider the incorporation of other GHG’s and additional sectors, where mitigation opportunities may be more readily available.

KEY WORDS: Sectoral ETS linkage, EPPA6, Brazil, Europe, alternative trading partners

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LIST OF ABBREVIATIONS

AAU	Assigned Amount Units
ABC	Plan Low Carbon Agriculture Plan
AEEI	Autonomous Energy Efficiency Improvement
BAU	Business as Usual
BC	Black carbon
BM&F	Stock exchange
BRA ETS	Brazilian Emissions Trading Scheme
BVRio	Green Stock Change of Rio de Janeiro
CCER	Chinese Certified Emissions Reductions
CDM	Clean Development Mechanism
CER	Certified Emission Reductions
CES	Constant Elasticity of Substitution
CFC	Chlorofluorocarbons
CGE	Computable General Equilibrium
CHN ETS	Chinese Emissions Trading Scheme
CH4	Methane
CO_2	Carbon dioxide
CO	Carbon monoxide
COP	Conference of the Parties
EC	European Commission
EJ	Exajoules
ELEC	Electricity sector
EINT	Energy intensive-industries
EPPA	Economic Projection and Policy Analysis
ERU	Emissions Reduction Units
ETS	Emissions Trading Scheme
EU	European Union
EUA	European Allowances
FGV	Getulio Vargas Foundation
GAMS	General Algebraic Modelling System

GDP	Gross Domestic Product
GHG	Greenhouse Gases
GTAP	Global Trade Analysis Project Version
GVCes	Centre for Sustainability Studies
GWP	Global Warming Potential
HFC	Hydrofluorocarbon
IEA	International Energy Agency
IP	Industrial Plan
IPCC	Intergovernmental Panel on Climate Change
Rio-92	Rio de Janeiro Earth Summit in 1992
ITMO	Internationally Transferred Mitigation Outcomes
JI	Joint Implementation
KAZ ETS	Kazakhstan Emissions Trading Scheme
KETS	Korean Emissions Trading Scheme
LULUCF	Land Use, Land-Use Change and Forestry
MAC	Marginal Abatement Cost
MBRE	Brazilian Market of Emissions Reductions
MCP	Mixed Complementarity Problem
MIT	Massachusetts Institute of Technology
MOU	Memorandum of Understanding
MPSGE	Mathematical Programming System for General Equilibrium
MRV	Monitoring, Reporting and Verification
MSR	Market Stability Reserve
NAP	National Allocation Plans
NBFC	New Brazilian Forest Code
NDC	National Determined Contribution
NER300	New Entrant Reserve 300
NGO	Non-governmental organisation
NH3	Ammonia
NOx	Nitric oxide and nitrogen dioxide
NZ ETS	New Zealand Emissions Trading Scheme
OC	Organic carbon

PFC	Perfluorocarbons
PMR	Partnership for Market Readiness
PNMC	National Policy for Climate Change
PPM	Parts per million
RD&D	Research, Development and Demonstration
REDD+	Reduce Emissions from Deforestation and Forest Degradation
RGGI	Regional Greenhouse Gas Initiative
SAM	Social Accounting Matrix
SF6	Sulfur hexafluoride
SO_2	Sulfur dioxide
UK	United Kingdom
UNFCCC	United Nations Framework Convention on Climate Change
USA	United States of America
VAT	Value-added Tax
VOC	Volatile organic compound
WCI	Western Climate Initiative

CONTENTS

CHAPTER 1
GENERAL INTRODUCTION

1.1 Context

Environmental problems such as global climate change have been recognised as the major challenge of modern society. Scientific evidence has underlined the irreversible planetary damage from anthropogenic action, which emerges from the long-term accumulation of Greenhouse Gases (GHG) emissions in the atmosphere, particularly carbon dioxide (CO_2). Since pre-industrial periods, carbon emissions are considered to be the major source of air pollutants, being responsible for the so-called global warming (Stavins, 1997a; Belini, 2005; Bollen *et al.*, 2009). Nevertheless, global emissions are continuously growing and tend to exacerbate adverse effects on ecosystems and economies in the long-term. More specifically, failure to maintain atmospheric concentrations of GHG below dangerous levels will deliver long-lasting consequences.

From the economic perspective, the climate change problem reflects the lack of market value or well-defined property rights associated with the atmosphere since it is an open-access resource. Given the limited capacity of the biosphere to absorb emissions, it is imperative to implement governing rules to make economic agents account for environmental costs. Because the global atmosphere is a public good, and there is an inherent free-riding stimulus, combating climate change requires effective international cooperation, which needs to be codified in a combination of strategic policy instruments such as command-and-control (regulatory) and market-based policies.

These mitigation policies have been at the centre of the international debate on climate change for several decades, but only recently have market-based approaches been considered. The climate policy architecture under the Paris Agreement envisions international cooperation to achieve significant progress on emissions mitigation, which will require more ambitious action in the future (Millar *et al.*, 2017). By including provisions for carbon pricing at international level to comply with Nationally Determined Contributions (NDC), the accord provides the foundation for a new architecture in which linkages of national climate policies play an important role. In this sense, linked ETS systems would be viewed as a precursor to, and a stimulus for, a global ETS approach.

The literature highlights potential opportunities and benefits from the use of market-based instruments to facilitate emissions reductions whilst enhancing economic performance, mostly via integration of Emissions Trading Schemes (ETS). These schemes are based on the principle of cap-and-trade. In a two-way ETS cooperation, emission allowances are mutually recognised for the purpose of compliance with the local cap. This type of international cooperation creates a price for pollution, which equates marginal abatement costs between regulated jurisdictions, thereby increasing the access to abatement options and making it possible to attain the proposed mitigation target at a minimum cost to society.

Those aggregated cost savings could be an important argument against political resistance to link ETS systems. Further, it could be used as an opportunity to incentivise jurisdictions to commit to targets that are more ambitious, and to agree upon a linkage that can also generate funds to re-invest in more reductions. This climate finance may be economically appealing for the price signal it provides to attract investments in sustainable infrastructure (Studart and Gallagher, 2016) or to promote clean technology investments and economic efficiency (Farid *et al.*, 2016; IETA, 2016), particularly for developing countries.

In light of that, developing countries are encouraged to also take action with the support of developed countries. In the past, developing countries had been involved in climate change mitigation through flexibility mechanisms, as hosts of Clean Development Mechanism (CDM) projects, or in some cases, by committing to voluntary reduce emissions. Recently, a nationwide Chinese ETS has been launched following some years of experience with subnational pilot markets. The Korean ETS has been operating since 2015 and the Kazakhstan ETS has just reinitiated operations.

Carbon trading is likely to become even more common post-2020, as further countries plan, or at least investigate the potential for ETS adoption. As a result, linkages have the potential to develop among participants in the future.

To date, a small number of the active national and subnational carbon markets are involved in, or are open to the concept of, ETS linkages. Examples include the California, Quebec and Ontario link (the Western Climate Initiative, WCI) and the Regional Greenhouse Gas Initiative (RGGI) in the northeast of the USA. The European Union Emissions Trading Scheme (EU ETS), the largest and most consolidated system in the world, displays willingness to link with other compatible systems, which means other ETS systems with similar environmental integrity and system architecture could potentially link. There is currently a Norway-EU linkage, which also regulates the aviation sector.

Some aspects need to be considered when deciding on linking, for instance, existing differences on the level of ambition, the ETS design and regulatory rules, potential domestic distributional impacts, and political support. Rather than enhance environmental effectiveness, the climate policy may be impaired where there are differences in the relative stringency of targets or the design features of the ETS differ among participants. Although engaging in linking demonstrates the effort to establish comparable caps and attract political support, it can also signalise that lower ambition is acceptable or that there is a loss of national regulatory control. Furthermore, distributional impacts associated with financial transfers from trading may be an additional issue. The ultimate success of linking will depend on the participating jurisdictions and the design of the arrangements in each link, since heterogeneity between partners may affect the outcomes.

Several studies have been carried out in order to evaluate linking with the EU ETS, including the possibility of linking with non-EU schemes such as South Korea, China, Australia and California. Some of these studies investigated the effects of sectoral ETS linkage under different circumstances. For instance, international trading was investigated among all developed regions and electricity sectors of developing regions in Hamdi-Cherif, Guivarch and Quirion (2011). Similarly, Gavard *et al.* (2011) designed a national US-ETS in a linkage with a hypothetical electricity ETS system in China. Gavard, Winchester and Paltsev (2016) modelled a sectoral ETS on electricity and energy-intensive industries in the EU, the US and China, simulating autarky and linkage scenarios. Hübler, Voigt and Löschel (2014) assessed a Chinese ETS regulating energy-intensive industries, electricity, heat, petroleum and coal products considering a potential cooperation with the EU ETS. Results from these studies showed an increased adoption of low carbon technologies, a lower international leakage and generally, a greater degree of acceptance from developing countries to participate in the carbon market set by developed countries.

The framework introduced in this thesis considers linkage implications of a hypothetical Brazilian ETS with a similar sectoral coverage to the aforementioned studies. Among developing countries, Brazil has been an early adopter of climate commitments from an international perspective. With approximately 4% of global emissions between 1990 and 2014 (SEEG, 2016), Brazil agreed to promote a cut of 37% and 43% of 2005 emissions levels by 2025 and 2030 respectively, in addition to a commitment to stop illegal deforestation.

In the proposed Brazilian NDC, significant emphasis is given to the reduction of emissions from land use change and deforestation, which contributed to 27.5% of total emissions in 2010. Further strategies are planned for the agriculture sector, as the share of emissions correspond to 32% of the Brazilian emissions profile (MCTI, 2014). Notwithstanding the relatively low carbon intensity of the energy mix, Brazil still relies on the production and consumption of fossil fuels, which have the potential to hinder a genuine mitigation towards sustainable levels. Energy and industrial sectors combined, contribute 36.3% to total emissions. Therefore, climate policies aimed at energy-related sectors are also required to help achieve national climate goals, for example, through carbon pricing.

The Brazilian government has been supporting, in association with the World Bank - Partnership for Market Readiness (PMR), a comprehensive group of studies based on carbon pricing for the post-2020 period. Despite that, Brazil has not yet defined or even decided on whether to implement a domestic ETS. The arrangements for market instruments in the Paris Agreement may encourage Brazil to design a carbon trading system. By taking the lead, Brazil may encounter new opportunities for climate cooperation with developed systems, with the EU ETS being a potential candidate. Alternatively, a bilateral sectoral ETS linkage with emerging schemes such as the Chinese ETS, or a hypothetical Latin American ETS may be envisioned.

The implications of such proposals have to date not been investigated to date, as carbon pricing and related linkages have just emerged as a reasonable alternative for developing countries. This is reflected by the late incorporation of climate issues into the Brazilian domestic agenda, that is, the secondary relevance given to environmental issues in light of other political national priorities. Additionally, it demonstrates that developing countries are envisioning environmental and economic opportunities from ETS systems. The expected benefits of accessing the market and joining a linkage are related to the exporter role developing countries would presumably assume (Somanathan, 2008). However, it is not evident whether or not linking sectoral ETS systems is economically and environmentally feasible or politically desirable from the developing country perspective.

1.2 Objectives

Overall, the aim of this thesis is to evaluate the effects of adopting market-based instruments, more specifically, a carbon mitigation and trading strategy in Brazil, from 2020-2050. In this study, the proposed ETS design is defined to mimic the EU ETS, serving as a realistic prototype for other planned systems. Evidence suggests that smaller systems tend to cede proprietary features, which could be the case for Brazil. In addition, such upfront harmonisation is important to facilitate the linkage negotiation, since subsequent adjustments are more difficult due to path dependencies in the ETS design and implementation, as well as the need to honour initial political commitments (Görlach, Mehling and Roberts, 2015). Hence, the sectoral ETS exclusively regulates CO_2 emissions from energy intensive industries and the electricity sector[1].

From the linkage point of view, it intends to measure the costs and benefits of a developed-developing region sectoral ETS linkage, i.e. a linkage between the EU ETS with a proposed non-EU scheme – a Brazilian ETS. If Brazil implements a nationwide sectoral ETS, new opportunities for climate cooperation may arise, including with emerging schemes in developing countries. Furthermore, we also evaluate a potential ETS policy proposal with Latin America and China. These linkage candidates are considered due to geographic proximity and historical economic relations, for example, via the trade of international goods and services.

To estimate the effects of climate cooperation, this two-fold analysis takes into account the emission reduction pledges of the Paris Agreement by 2030 and for modeling purposes, projects the mitigation target to rise over the period so that Brazil, Europe, Latin America (and Mexico) achieve a 50%, 73% and 35% (50%) emission reduction by 2050, respectively, whereas abatement represents an 80% reduction of GDP intensity in the same year for China.

The appropriateness of this climate strategy for helping participating jurisdictions to achieve a low carbon economy, as well as the degree of compatibility among proposed trading options, is also evaluated. Firstly, the study identifies the heterogeneity of the proposed partners (Brazil, EU, Latin America and China) in terms of their macroeconomic profile and energy mix, highlighting sectoral emissions levels and the stringency of targets. This allows characterisation of the extent to which a proposed partnership may be recommended.

The second part discusses the feasibility and compatibility of the policy proposal, with the aim of understanding to what degree international coordination via an ETS may cope with several political challenges without undermining sustainable development. In fact, gathering multiple actors that

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[1] For modelling purposes, other important features of the EU ETS are not included in this approximation, such as the availability of offsets or the inclusion of the aviation sector. The phases of compliance are defined according to the periodization of the model, that is, in 5 year-intervals.

may diverge in their perceptions of costs and benefits from climate action can render the linkage to be perceived as unsuitable for respective economic and environmental interests. An appropriate level of ambition is therefore fundamental from each party. In short, the thesis is developed to address the following questions:

- How significant are the costs to reduce emissions in Brazil via a sectoral ETS?
- Does the modelled ETS have the most appropriate structure for Brazil?
- Under what conditions is linking more advantageous or disadvantageous than a domestic sectoral ETS?
- What is the most appropriate trading partner for agreeing on a bilateral climate deal with Brazil?
- What are the major implications of each linkage in relation to factors such as carbon price, level of emissions, GDP, welfare, sectoral output, international trade, revenues from trading, and energy substitution towards low-carbon technologies?
- What is the trade pattern in each proposed linkage?

To answer these questions, this quantitative exercise employs the MIT computational general equilibrium (CGE) model, the Economic Projection and Policy Analysis in its sixth version (EPPA6). The CGE approach is useful for such policy analysis since it is able to translate the carbon constraint into economic responses of agents across multiple sectors and regions. Moreover, it exerts a powerful influence on climate policy-making decisions by pointing out long-term tendencies to anticipate potential outcomes comparatively. Economic modelling is rather necessary in the context of this thesis, as most of the economic indicators of ETS linkage cannot be quantified empirically.

Hence, the ex-ante analysis developed here is especially important for the policy recommendations that emerge, whereby this study fills the existing gap in the literature. Specifically, it contributes to the literature of applied policy analysis with emphasis on developing countries and climate agreements via sectoral ETS linkages. Given that those countries have to date not been involved in any direct linkage, but ETS implementation is increasingly taking shape around the world, the evidence from this quantitative exercise is relevant for potential future negotiations. For Brazil, an emissions mitigation and trading strategy could encourage a more aggressive contribution to address climate change, thereby incentivising other developing countries, including the largest economies, to curb emissions.

Compared to previous simulations, the study explicitly focuses on the most cost-effective ETS design for a linkage where Brazil participates, distinguished according to the stringency of Brazilian targets, the use of flexibility provisions, the recycling of revenues to alleviate distributional impacts, and the participation of other regions through ETS commitments. The core scenarios are considered from three perspectives: i) a situation where there is no mitigation policy applied, ii) a domestic ETS[2] and iii) a linked ETS. Among the criteria to assess the implications of a linkage proposal between Brazil and another developing country (China), a regional organisation of developing countries (Latin America) or a developed-world programme (EU ETS)[3], one can mention the impacts on emissions and energy or the costs of abatement as well as the distributional effects.

1.3 Dissertation Structure

The thesis is organised in six chapters, including this general introduction – Chapter 1. Chapter 2 consists of a theoretical literature review of the economics of climate change, with emphasis on policy instruments for emissions mitigation. The chapter presents the climate change problem and discusses the relevance and potential benefits of pricing carbon, particularly via

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[2] The focus of this research is to analyse alternatives for Brazil in terms of a bilateral climate agreement rather than an isolated sectoral ETS. In reality, however, the majority of ETS linkages were negotiated after some time of experience. In other words, it is highly likely that a domestic ETS precedes the linkage. To look into this in depth, Appendix B discusses different scenarios for the Brazilian ETS.

[3] Literature highlights that a sectoral ETS approach is less efficient than a global ETS system, even though it could lead to a global agreement. Assuming that a global ETS comprising all economic sectors is rather feasible in the short-term, it is included an additional simulation for a global sectoral ETS and related effects in Appendix D.

cap-and-trade. The challenges of addressing climate change are framed from an international cooperation perspective, i.e. the prospects and challenges related to linking ETS systems. Chapter 3 sheds light on the role of international negotiations towards a Post-2020 world of ETS linkages, summarising experiences with linkage to date. In particular, it explores the Brazilian and European case. Chapter 4 introduces the CGE modelling framework and scenarios considered for simulations with EPPA6. The main results and discussions are exhibited in Chapter 5. Finally, Chapter 6 offers conclusions and policy implications, as well as underlining recommendations for future studies.

CHAPTER 2

ECONOMICS OF CLIMATE CHANGE: AN OVERVIEW ON POLICY INSTRUMENTS AND INTERNATIONAL COOPERATION FOR EMISSIONS MITIGATION THROUGH CARBON PRICING

2.1 Introduction

Global climate change is one of the most serious and urgent issues of the 21st century. There is increasingly compelling scientific evidence highlighting the irreversible planetary damage from anthropogenic action (Goulder and Pizer, 2006; Commission on the Economy, 2014; Edenhofer *et al.*, 2015; Harris, Roach and Codur, 2017). This environmental problem emerges from the long-term accumulation of Greenhouse Gases (GHG) emissions in the atmosphere, particularly carbon dioxide (CO_2). Since pre-industrial periods, carbon emissions are considered to be the major source of air pollutants, being responsible for the phenomenon now commonly referred to as global warming (Stavins, 1997a; Belini, 2005; Bollen *et al.*, 2009)[4].

Overall, CO_2 and other Greenhouse Gases are released into the atmosphere from many activities including resource extraction, deforestation, energy generation from fossil-fuels, industrial processes, transport, agriculture and waste management. In fact, virtually all aspects of economic activity, including individual consumption, business investment and government spending increase the level of emissions and hence, affect the global climate (Aldy and Stavins, 2011; Baranzini *et al.*, 2017).

Over the last decades, economic growth around the world, especially in developed countries, has contributed to a rapid increase in the level of GHG concentration, which has doubled since the beginning of the 1970s (WTO-

[4] Other sources are also important contributors to the global warming effect, measured by the Global Warming Potential (GWP). This is an index for measuring the amount of warming effect a gas causes compared to carbon dioxide, which has the index value of 1. Thus, a larger GWP indicates a greater potential to warm the planet in a given period of time, usually 100 years. Methane (CH4) and Nitrogen oxides (NOx) have the lowest GWP, the former with 25 and the latter with 298. This indicates, for example, that the one ton of methane emitted has the equivalent impact of 25 tons of carbon released in to the atmosphere over 100 years (BRADER, 2012). In general, other GHG's present higher GWPs, namely chlorofluorocarbons (CFCs), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulfur hexafluoride (SF6).

UNEP, 2009). This is due to the fact that GHG emissions are related to the existing patterns of production and consumption. The effects of such industrialisation on the climate system as a whole have not been regarded (Costanza *et al.*, 2014).

The science of climate change explains that GHG emissions present a long-lived characteristic, that is, they remain in the atmosphere for long periods, being very difficult to reduce (Commission on the Economy, 2014; Harrod and Martin, 2014). As a result, they continue to affect the global climate long after being emitted in a nearly uniform distribution. This is independent from where emissions were released since they expand through the atmosphere across political borders. Whilst the emitter country faces no cost for the pollution generated, all others are affected.

Observed changes in the environment are, therefore, directly influenced by the economic development of last century. This is due to the economy-environment linkages, which involve a circular flow of materials and energy extracted from the environment, and which are used as input to the production of goods and services which are available subsequently to consumers. As a result, economic processes generate outputs in the form of residuals, flowing back into the ecosystem through environmental receptors, such as the air system (Perman *et al.*, 2011). Indeed, there are mutual implications from imbalances between these inflows and outflows of the global economy[5].

In the context of an increasingly globalised world, future emissions pathways are not sustainable if stringent constraints are not put in place. For comparison, scientific projections have already demonstrated that, in terms of atmospheric concentrations, the world has already surpassed the safe threshold of 400 parts per million (ppm) of CO_2 or around 450 ppm of CO_2eq, the level above which the risk of climate imbalance is imminent (WTO-UNEP, 2009; IPCC, 2014). In the next decades, it will be very challenging to hold the level of emissions consistent with the probability of keeping global warming below 2°C compared to pre-industrial levels by 2100, as estimated by IPCC (2014),

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[5] The environment plays a fundamental role for the economy, from being a supplier of resources and provider of services for global life, to a sink for waste.

especially because GHG have been accumulating an additional 2 ppm per year in the atmosphere.

Although the severity of climate change effects are not consensually predicted in the climate science community (Dryzek, Norgaard and Schlosberg, 2012), the noted perspective has the potential to intensify the phenomena, causing extreme weather events and temperatures, modifications in the precipitation pattern, sea-level rise and greater occurrence of desertification and flooding (Gunningham and Sinclair, 1998; Goulder and Pizer, 2006; Harris, Roach and Codur, 2017). These biophysical changes are detrimental to the functioning of ecosystems, the viability of wildlife, and the well-being of humans.

However, IPCC (2014) and Stern (2006) point out that it tends to be more severe in the most vulnerable group, i.e. the developing countries, both in environmental and economic terms. Vulnerability is determined by exposure, sensitivity and ability to adapt to climate change (IPCC, 2014). The disproportionate exposure to climate change of poorer, hotter, and lower-lying countries configures another motive to find ways to avoid GHGs (Tol, 2018).

To date, these countries have contributed the least, even though current estimates indicate that a significantly higher proportion of emissions will arise from these countries in the coming decades (WTO-UNEP, 2009). Further development efforts to address poverty, by means of reducing these effects are strategic, especially because they can deeply trap the poorer in this condition as a result of lower growth rates of the economy. For Costanza *et al.* (2014), there are significant advantages in abandoning the fossil fuel model, and industrial countries play an important role in assisting developing countries towards qualitative development and the subsiding of alternatives.

The challenges of climate change can be associated to the fact that it is globally caused, and a public good case[6]. Contrary to other commodities, the atmosphere is an open-access resource with no legal limitation on the usage level, i.e. the amount of GHG released, so that it transcends a conceivable scale. In the absence of well-defined property rights, economic principles for its

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[6] Many other environmental resources are public goods, for example, water quality or biodiversity.

use differ from the rules applied to private goods that are traded in the market since the price system is unable to properly value the atmosphere (Seroa da Motta, Ruitenbeek and Huber, 1996).

When natural resources are collectively utilised without governing rules or a market value, there is a tendency for overexploitation. In the literature, it is known as the "Tragedy of the Commons", popularised by Garret Hardin (1968). Simply put, the concept refers to the problem of open access of common resources. Whilst economic theory advocates that market forces efficiently allocate the resources, it acknowledges that the market fails to be efficient if the public resources are overused, negatively impacting the environment – the tragedy. However, economic agents will continuously degrade the common or shared resource (the climate) they benefit from, unless there is coercion through regulation or a stimulus through adequate allocation of property rights.

This interpretation of society-nature interaction is fundamental to orientate around the assumed biophysical limits and the economy (Bresnihan, 2017). For effective management, the common resource must be accounted for in line with these limits, even if uneven allocation is necessary, as noted in Hardin (1968). In the climate case, the dilemma lies in challenging the economic interests of each country and understanding the finite nature of the global common, particularly due to an inherent free-riding behaviour.

When it comes to climate change impacts, there are no national boundaries, but still countries are typically reluctant to voluntarily incur the private costs for public benefits of a stable climate. Since mitigation is non-rival and non-excludable, that is, the global climate is equally available for all agents without preventing from consumption those who do not pay for its protection, the benefits of emissions reduction in one country are shared by the entire world. As such, there are strong incentives to free-ride and to understate the willingness to pay[7].

At the same time, it is of interest to all involved to collectively promote mitigation for the common good. For effective international cooperation, there is

[7] If rules are to be applied, it is important to know the preferences of relevant agents in terms of marginal willingness to pay.

a set of strategic policy instruments available, such as command-and-control (regulatory) and market-based policies. If a global commitment is not successfully achieved, additional problems may emerge. An example is the concept of “carbon leakage”, which refers to incentives for polluting activities to move to jurisdictions with less stringent climate regulation, where production costs do not account for emissions reductions. However, international coordination at global level is very challenging and difficult to achieve as environmental policies requires enforcement among nations.

Another difficulty when coping with climate change is that certainty with regard to outcomes, i.e. future risks and damages, is not guaranteed. In fact, the complexity of the climate process, as a geo-atmospheric-ecological system, entails policy-making under large-scientific and economic uncertainties (Dryzek, Norgaard and Schlosberg, 2012; Commission on the Economy, 2014; Harrod and Martin, 2014). Of course, the existence of uncertainties does not imply inaction, particularly in light of the projected significant socioeconomic costs of climate change.

From an economic perspective, there are distinct effects of a changed climate since it may modify agriculture patterns and energy use, among other factors of economic activity. In Tol (2015), initial impacts are estimated to be positive and only in the long-run negative effects predominate. These impacts are usually differentiated according to the existing productive structure. For example, where the agriculture sector prevails, the production would be largely affected with loss of land area, disruption of water supply or input productivity, faster growing of crops due to carbon dioxide fertilisation, and loss of agricultural output caused by droughts or extreme cold weather. As for the energy sector, lower availability of resources may increase production costs whilst reducing efficiency.

In the presence of climate change, a synthesis of various results reported by IPCC (2014) predicts a global aggregate loss of 2% income if temperature increases by 2°C. This is consistent with a comprehensive literature review in (Tol, 2015), which suggests an overall impact on the economy and human welfare of approximately 1.3% in the context of 2.5°C global warming. This is considered to be a limited effect in the short to medium term, but there is a reversal tendency by the end of the 21st century, with larger negative implications.

Under this less optimistic outcome, decision-makers tend to be risk-averse. The rationality is to undertake measures for managing the potential risks of catastrophic implications of climate change in the present, rather than postpone to future generations. In this context, mitigation and adaption are two strategies necessary to neutralise GHG emissions.

Adaptation refers to adjustments made in natural and human systems towards actual or expected climate effects. The practices aim to soften the consequences of climate change while they also seize beneficial opportunities that emerge. Potential adaptation options available include adopting new technologies for building better infrastructure, whether to protect coastal areas against sea-level rise or to withstand hotter temperatures; individual shifts on behaviour by modifying food, transport or resource use choices; altered practices in economic sectors, such as in farming with different crops being cultivated or increased productivity; and finally, the introduction of planning regulation to help strengthen resilience to climate change through adaption.

On the other hand, mitigation is seen as a self-protection strategy. By promoting emissions reductions in order to limit the likelihood and the magnitude of climate change, it addresses the main cause of the problem. Broadly speaking, mitigation is an intervention that aims to lower the concentration of GHGs in the atmosphere, and does so by stimulating changes in resource-use decisions, including strategic investments in infrastructure, as well as low-emissions innovation. However, many decisions with respect to unsustainable behaviour about the use of environmental services are not reversible. As stated in Perman *et al.* (2011), the integration of irreversibility and imperfect knowledge of the future is an appropriate condition to keep mitigation options opened and to behave in a relatively cautious manner.

Even though a combination of both approaches is necessary, a particular emphasis is given to mitigation policies, as is also the case in this research. In reality, the more the inertia in introducing these measures, the more inevitable the potential for a 4°C rise in global temperatures this century. Indeed, the window of opportunity to avert climate change is rapidly shrinking (Gunningham and Sinclair, 1998), mostly because there is not much space for further emissions in the atmosphere, around 900 $GtCO_2$ which is equivalent to less than 25 years of emissions at the 2014 level (Edenhofer *et al.*, 2015; UNCTAD, 2016).

The literature is well established in this regard, underlining the need for immediate reduction of the rate of growth of anthropogenic emissions to avoid or attenuate ecological degradation (Stern, 2006; Flachsland *et al.*, 2008a; Perman *et al.*, 2011; Commission on the Economy, 2014), and indicating the costs of delayed participation in international collaborative efforts for mitigation (Jakob *et al.*, 2012).

For an immediate reduction in emissions to materialise, the consensus is that adequate collaborative policies are required where public stakeholders, institutions, private corporations, the scientific community and individual citizens participate. The so-called “common institutions” are increasingly being agreed or implemented worldwide: at community, regional, national or global levels. Unfortunately, considerable disagreement still exists in terms of what climate policies, if any, should be introduced (Goulder and Pizer, 2006), an often-common problem of collective public goods as previously discussed.

On the other hand, the fact that major emitters become subject to domestic and international policies signalises their willingness to collaborate for the global common good. Further, it encourages developing countries to take part, as indicated by the higher interest in implementing mitigation policies. Accordingly, initiatives integrating developed and developing regions in a cooperative approach play an important role. This is the approach of the recently ratified Paris Agreement set up by United Nations Framework Convention on Climate Change (UNFCCC). Although the future architecture of the agreement remains uncertain, it has made progress towards gathering countries to take climate action, “a first step to transform a free-access atmosphere into a common-property resource” (Perman *et al.*, 2011), or to transform carbon-based economies into low-carbon economies.

This chapter sheds light on the economics of climate change and the political instruments available for policymakers to correct market imperfections and help mitigation. Classical instruments of environmental policy are classified into regulatory or economic instruments (Seroa da Motta, Ruitenbeek and Huber, 1996; Almeida, 1997; Rathmann, 2012).

Yet, it is acknowledged in the literature that an effective policy would change how decisions of economic activities are made in order to encourage changes in energy production and consumption patterns towards a low carbon economy. For that, imposing a cost on the polluters is the most economically efficient manner to, at least partially, neutralise GHG emissions. For Nordhaus (2008) a well-designed policy is fundamental for a long-term low-carbon global economy, which also creates the right incentives for firms to develop new technologies. Based on that, the major recommendation is to implement price-type approaches, since these could be "powerful tools for coordinating policies and slowing global warming" (Nordhaus, 2008, p.164).

For the purpose of this thesis, this introduction discussed the rationale of the climate change problem. Section 2.2 provides a brief description of market-based instruments compared to regulatory policies. In recent years, the debate on climate cooperation through carbon pricing mechanisms has evolved, with linking of ETS system as one approach especially supported by the EU Commission (European Comission, 2009; Mehling and Haites, 2009). The basic concept of linking, and the economics behind it are introduced in Section 2.3, followed by a discussion of the pros and cons of integrating ETS systems.

2.2 A review on policy instruments for mitigation

There is extensive theoretical literature on the analysis of climate policies and the effects on the economy and society. In order to provide insights on how to address environment-related problems through efficient policies, reference is made to the economic theory concept of market failure in terms of externalities, as well as to private and social costs. As such, policy proposals aim to establish mechanisms to make economic agents account for environmental costs.

In the framework of economic analysis, GHG emissions and their effects configure a typical case of environmental externalities and overuse of a common property resource, as previous defined (Harris, Roach and Codur, 2017). Climate change is itself the biggest of all negative externalities (Tol, 2009; Commission on the Economy, 2014), "negative because they detract from social welfare and externalities because they are external to any accounting within the economic system" (UNCTAD, 2016, p. 64).

It is also dynamic, as the current level of pollution reduces the climate stability and alters its future conditions. As such, it represents an example of market failure due to the lack of a price signal to express scarcity for certain scarce environmental resources such as clean air or water (Taschini, 2009). Since market prices fail to internalise climate-related damages, the prevailing reliance on fossil fuels and the current mix of energy-consuming technologies continuously imply high levels of emissions into the atmosphere.

For the traditional economic theory, an externality is a side effect, whether positive or negative[8], of production and consumption decisions of one agent over the output (profit) and well-being (utility) of those not directly involved with the activity in uncompensated ways. Since these are unintended effects, public and private agents do not take them into consideration in the decision-making process. As a result, no reward or penalty is applied to the responsible agent.

[8] Classification of externalities is not limited to positive and negative effects. One can also specify them according to the economic activity where they originated (production or consumption), and by the economic activity the externality affects (production or consumption). For detailed information, see Perman *et al.* (2011).

In the presence of externalities, the market fails to give the appropriate signals to economic agents thereby reducing the capacity to allocate resources efficiently. This occurs because emitters do not bear the costs of adverse effects to society (Marshall, 1890; Pigou, 1920), implying a divergence between private and social costs (Jorgenson *et al.*, 2008).

From the climate change perspective, pollution emitted in one country at no costs expands through the atmosphere across political borders and affects all others. As long as economic agents do not factor costs into private-cost-benefits decisions (Oates and Baumol, 1988; Nordhaus and Boyer, 1999), the tendency is to continuously emit this pollution. Given the limited capacity of the biosphere to absorb emissions, it is imperative to take proper account of this market failure.

For that purpose, the literature calls upon the precautionary polluter-pays principle. The rationale is that social external costs, namely pollution abatement, costs of environmental recovery and compensation costs for victims of damages, if any, should be internalised in the decision-making process of polluter firms to guarantee the enforcement (OCDE, 1989; Mountondo, 1999). There are multiple climate benefits of correcting market failures, for instance, by incentivising polluters to undertake environmentally-friendly measures in order to neutralise the externality, which positively impacts human health by improving local air quality (and reduces the use of fossil fuels), as well as positive impacts for social welfare.

In the economic context, climate change can be addressed by a policy intervention to correct the market failure. This is typically a case for government intervention (Seroa da Motta, 2004; Averchenkova and Bassi, 2016), as it has the legitimacy and authority to create and maintain appropriate institutional arrangements to establish and support climate-related property rights (Perman *et al.*, 2011).

The classical paradigm for environmental policies lies in the regulator’s role in controlling private agents through regulation. The options available to internalise the external environmental cost are often classified into the so-called command-and-control (regulatory) or market-based (i.e. use of economic measures) instruments. By employing appropriate incentives on private behaviour, the intervention should provide the signal to limit the level of emissions, and hence, protect environmental quality. Changes in producer and consumer behaviour are driven by binding obligations or restrictions through regulation in the command-and-control approach, whereas economic instruments operate by creating incentives via prices and markets, for agents to make behaviour less polluting.

This conceptual framework forms the principles behind most public policies focused on environmental goals, even though they are not always effective or eventually fail to achieve desired outcomes if poorly-designed, incurring substantial costs. For the climate case, two main questions arise as to the optimal emission reduction target and the best method to effectively achieve it. Of course, the choice depends on the objective that is being sought, but also on the characteristic of pollution under consideration, whether it arises from the flow of the pollutant (i.e. the rate of emissions) or from the stock (i.e. the concentration rate).

For Barrett and Moreno-Cruz (2015) and UNCTAD (2016), in order to stabilise GHG concentrations, policies should aim to reduce the flow of anthropogenic carbon emissions or prevent them from reaching the atmosphere, and/or focus on removing the concentrations of these gases from the atmosphere[9]. Flow and stock pollution requires, in this sense, different policy categories to meet the proposed target of mitigation.

In practice, mitigation levels are frequently set up on the grounds of economic efficiency[10] or sustainability[11]; but mostly as a result of pressures

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[9] There are technologies available to capture and store carbon emissions before they reach the atmosphere and to promote industrial air capture.

[10] Traditional economic theory describes the rationale of the economic agent on the notion of profit maximisation at the least cost. The variable "price" is, therefore, very important and determines the efficient resource allocation; so agents and price behaviour are aligned. In the climate context, economic efficiency is related to policies in which resource allocation maximises social welfare. The use of efficiency to evaluate how much pollution there should be was introduced by Pigou (1920) from the concept of externalities. An optimal level of pollution is generated, enabling societal benefits to exceed the costs. This is an equivalent outcome to the pollution being fully internalised. It is argued that a zero level of pollution is desirable, however, it is not economically efficient since a certain level of pollution is always necessary to produce goods and services (Van Beers and Van den Bergh, 1997; Perman *et al.*, 2011).

from various countries and stakeholders, which ultimately involve the acceptance of public opinion or the political feasibility – sometimes defined by the interplay of pressure and sectional interests (Perman *et al.*, 2011). At the global level, Nordhaus (2008) states that optimal mitigation requires equalising the incremental or marginal costs of reducing emissions in each sector and country, but the world is not even close to moving towards this direction.

More than only weighing the economic costs against damages of climate change, nations usually opt for a certain level of emissions reductions on the basis of forthcoming net benefits, such as international transfers. This particularly holds for those implementing stringent climate policies (Stavins, 1997a), or to developing countries in which coping with mitigation commitments may derail the immediate objective of rapid economic growth and poverty reduction (Commission on the Economy, 2014).

For each case, there is a wide range of policy options (and associated costs) available to environmental regulators to control emissions release into the atmosphere. According to the circumstance, the use of specific instruments is favoured, depending on the set of attributes it has and how well they match with the objectives pursued. It is beyond the scope of this thesis to discuss the vast amount of literature related to these instruments. Instead, a brief overview will be provided in order to compare instruments whose objective is to curb emissions, and gather efforts towards a deep decarbonisation of the global economy.

The various approaches to environmental policy introduced in this chapter are thought as a set of targets and instruments that seek to reduce the negative impacts of human activity on the environment (Nascimento, Nascimento and Bellen, 2013). These policy proposals are based on the concept of externalities, private and social costs. The extent to which each of these influence decision-making, has the potential to prevent the emergence of

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[11] Sustainability has different definitions in the literature. To simplify, it is considered here as meeting the current needs of the population by improving living standards without compromising those of future generations. In these terms, sustainable development refers to economic conditions that generate employment opportunities, being socially equitable and inclusive, ecologically balanced and adapted to climate change impacts (Yu, 2009; Romeiro, 2012).

externalities or to help internalise existing ones, combining ways to reward or to punish those who do not respect the rules.

These mitigation policies have been at the centre of international debate on the climate change agenda for several decades, but only recently has there been a broader use of them. The appropriate strategy is context-specific and subject to political and economic constraints. Among the criteria to assess the appropriateness of instruments are the impacts on income, wealth distribution, the cost of abatement, and the likelihood to reduce the degree of negative effects on the environment. Stavins, Kennedy and Newman (1997) distinguish between domestic and international strategies and affirm that successful experience with targets specified domestically prevails over the adoption of initiatives in a joint framework (bilateral, multilateral or global).

On the global governance side, an attempt to combine a mix of policies within a new cooperative architecture among nations is envisioned post-2020. A key consideration on the choice of the policy instrument to be investigated in this study is the incentive to change behaviour it can provide, by putting a price on pollution. This is especially the case for regulated sectors and jurisdictions, which are incentivised to invest in new technologies or to adopt alternative, low-emission technologies. The theoretical and empirical framework of the literature below compares the method of operation, and relative advantages of the instruments.

2.2.1. Defining regulatory (command-and-control) and economic (market-based) instruments

Among the strategies for environmental regulation, the historically dominant method of reducing pollution is to impose direct control with regard to the quantity of emissions discharged. This set of control is commonly known as the command-and-control measure, which is the most conventional due its political appeal (Pizer, 2002) and has been adopted by virtually all countries for dealing with environmental emissions (Stavins, 1997b; Gunningham and Sinclair, 1998; Taschini, 2011).

This instrument refers to climate policies based on direct regulatory intervention for endorsing behaviour change by imposing obligations, which allow little flexibility in the means of achieving the goals. The political authority specifies the actions to undertake in order to achieve environmental objectives, as well as the technologies or products to be used (or prohibited) in the production process. Standards may be specified based on acceptable and unacceptable behaviour for economic sectors (Goodstein, 2010; IPCC, 2014). It additionally defines the type of enforcement machinery to ensure compliance with the law (Görlach, 2013).

The literature associates the advantages of this regulatory instrument with the relative certainty over the environmental outcome. It also highlights that the main peculiarity of this approach is the legal treatment given to the polluter inasmuch as it has to comply with the specified rules, otherwise it would be subject to penalties in judicial or administrative proceedings.

The regulation imposes identical pollution-control burdens on organisations, irrespective of differences in relative costs (Stavins, 1997a). For this purpose, standards are mostly used, being usually distinguished into performance or technological requirements. Stavins (1997a) states that this approach is useful in the context of GHG emissions, where governments could use it to ban or attempt to alter the use of materials and equipment considered as harmful for the environment. For instance, whilst performance-based standards mandate specific outcomes per unit of product, technology-based ones specify technologies for abating pollution or production methods to comply with the regulation.

In the case of performance standards, the permissible level of emissions discharged, is determined with respect to the actual organisational situation. There is therefore discretion concerning the choice of technical and other solutions used to achieve these levels. In contrast, technology standards stipulate particular equipment, processes or procedures to control emissions and improve energy efficiency[12].

This type of regulation has been broadly used worldwide and it is especially relevant for countries where consumers' energy awareness is low due to a historically low energy price (IPCC, 2014). Overall, it has served as a basis for regulators unfamiliar with any environmental policy. Literature shows that previous experience with a regulatory instrument may provide a foundation for future market-based mechanisms by creating the institutional capacity in policy evaluation, monitoring and enforcement (Russell. and Powell, 1996; Legro *et al.*, 1999). Among the advantages, one can mention the simplicity, familiarity and acceptance by major emitters and interested groups, enhancing the support of society. Along with clear environmental goals, these aspects are necessary for a well-functioning regulatory system (Costanza *et al.*, 2014).

Notwithstanding the relative effectiveness in delivering a particular emission-reduction objective, the disadvantage is that command-and-control methods have been generating non-effective (or counterproductive) results on organisations, which end up using expensive, and sometimes inappropriate, means to reduce emissions (Stavins, 1997b; Aldy and Stavins, 2011). Another problem commonly mentioned stems from inadequate monitoring and enforcement activities, with potential to compromise the achievement of policy goals.

Moreover, direct regulation has provided limited dynamic incentive for polluters to search for climate-friendly and economically better technologies, particularly in the long-term, given that it is limited to meeting the minimum

[12] Another possibility to enforce a regulatory system is to forbid or restrict activities during certain periods or in certain areas (Almeida, 1997).

requirements, and may not require significant additional efforts. Ultimately, it may not induce innovation and technological change that might result in greater levels of emissions reduction (Jaffe, Newell and Stavins, 2003; Sterner, 2003)[13]. In short, the disadvantages of direct regulation involve the low potential for meeting environmental quality cost effectively[14]. Most importantly, there is little financial incentive for firms to exceed the emissions target stipulated by the regulator.

At the same time, the regulatory approach has had limited success in achieving the desirable level of emissions reductions. Figure 1 reveals that over the last decades tackling global emissions through regulation has hardly produced significant pollution control. This is particularly true for developed countries such as Europe (Costanza *et al.*, 2014) in which carbon emissions dropped only approximately 17% from 1990 to 2014, being also influenced by the economic crisis in the 2000's. Whilst global emissions increased by 61%, carbon emissions in Brazil grew 142% in the same period.

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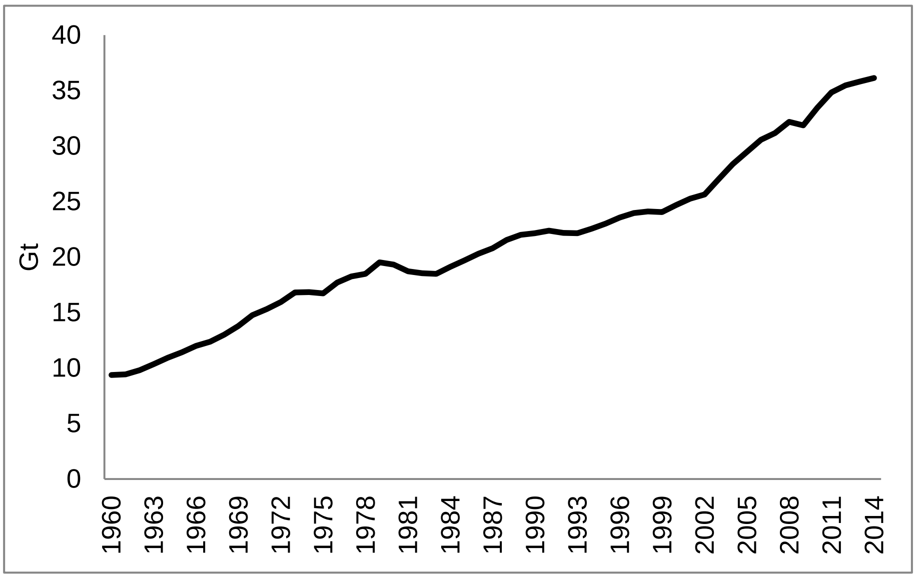
[13] A technology mandate may to some extent influence technological change however, determining the cost-effectiveness is a very difficult task for regulators. In this context, Jaffe, Newell and Stavins (2003) argue that either costly overly stringent, or alternatively, low ambitious requirements may be implemented.

[14] Based on Almeida (1997), Goulder and Parry (2008), Costanza *et al.* (2014), disadvantages can be summarised as follows: i) high level of administrative costs (enforcement and monitoring) that require extensive resources and also originate high expenditure per unit of pollution reduction; ii) deduction of cost-effectiveness since marginal cost structures differ from one source to another: the more marginal cost varies between sources, the less effective a regulation might be; iii) creation of barriers to entry and then, the existing market structure tend to be perpetuated; iv) lack of economic stimulus for diminishing pollution by more than is required: it limits the incentives for technological development and pollution prevention; v) lack of acknowledgement of the costs imposed by polluters upon society at the time a decision is made.

Figure 1 – Global CO_2 emissions from 1960 to 2014



Source: World Bank (2018).

From this evidence, and given the ubiquitous nature of GHG emissions as well as the urgent need to deeply decarbonise the global economy, it is rather unlikely that a centre-piece of a meaningful climate policy will be based solely on this mechanism. Instead, it calls for alternative approaches to address climate change (Goodstein, 2010; Rathmann, 2012). To cope with inefficiency and other issues from the use of regulation for addressing environmental problems, the literature (Philibert and Reinaud, 2004; UNEP, 2004; Stern, 2006; Costanza *et al.*, 2014; UNCTAD, 2016) suggests it is useful to combine economic incentives and a regulatory system, rather than just replace them, so as to "reduce or eliminate emissions of concern, and to shift consumption and production patterns towards greater sustainability" (UNEP, 2004, p. 19).

This strategy seems to be consensually advised in order to enhance the effectiveness of the climate policy, which is a metric that allows "researchers and politicians to rank mitigation policies" (Feld and Galiani, 2015, p. 10), and to allow a more rapid and extensive policy response.

Economic instruments, or market-based policies, have been increasingly drawing the attention of policymakers due to some advantages they offer over regulatory instruments in terms of mitigation, especially the inherent financial motivation. In fact, a market-based instrument can be applied to comply with a certain regulation (Pearce and Turner, 1990). This instrument has long been advocated as a more efficient and less costly manner of dealing with environmental management (Oates and Baumol, 1988), which can largely benefit the countries that adopt them (UNEP, 2004). Cost-effectiveness is considered a desirable and efficient attribute to allocate the smallest amount of resources to control pollution while realising a given level of emissions reductions (Pizer, 2002; Perman *et al.*, 2011).

If properly designed and implemented, a climate policy based on economic instruments allows the least overall cost to society, since those organisations that can promote deeper reductions more cheaply are economically encouraged to do so. This occurs because the costs of achieving a certain environmental quality can be minimised, by maximising flexibility of response, as the emitter is able to choose the level of production and corresponding level of emissions being determined by the market (Seroa da Motta, 1996; The Royal Society, 2002).

Thus, whilst ensuring that economic actors pay a price for each unit of their emissions, internalising (part of) the climate-related cost (Stavins, 1997b; Perman *et al.*, 2011; UNCTAD, 2016) and affecting the demand for this resource, provides regulated entities flexibility to define the most appropriate method and technological alternative through cost and benefit assessments (Tomas and Callan, 2010; Taschini, 2011; Moarif and Rastogi, 2012).

Hence, by altering the price signals, employing market incentives and penalising or rewarding regulated industries, this mechanism incentivises a shift towards less polluting behaviour. Within a market structure, regulated sectors and related organisations tend to engage in activities intensive in emissions unless confronting a price that equals to the marginal cost of reducing their pollutant discharges (Taschini, 2009; Goulder and Parry, 2008). The relative cost-effectiveness depends on the existence of significant differences in abatement cost among polluters (Sterner, 2003).

In this circumstance, a mitigation target is achieved at minimum cost when the incremental amount that organisations spend to reduce emissions, the marginal abatement costs, are equalised across economic actors (including households), through a common price. The lower unit cost of abatement guarantees efficiency and effectiveness over traditional regulation. However, the cost of emission control and the price paid for emissions are often passed forward into the prices of final goods and services.

Consequently, the policy indirectly affects consumers' behaviour since they confront higher prices reflecting the emissions price. Yet, Harrington, Morgenstern and Sterner (2004) argue that, as opposed to the supply side, there is no corresponding mechanism to limit the use of environmental services on the demand side because no autonomous adjustment in prices is available to maximise all individual's satisfaction.

Most of climate policy theoretical literature highlights the potential opportunities of adhering to an incentive-based policy intervention on the basis of environmental effectiveness, another relevant criterion for evaluating its benefits (Anderson, 2002; Rydge, 2015). This concept can be described as the potential for meeting the environmental objectives, that is, to deeply reduce the level of emissions. A well-designed market-based mitigation policy prices the unit of emissions, giving a clear and credible signal to guide expectations in business. As such, it contributes to dynamic efficiency because it continuously stimulates further emissions reductions, as a response of flexibility with the aim of achieving those reductions. In light of the long-lasting accumulation of emissions in the atmosphere, a dynamic long-term incentive is required to influence economic and technological change (Feld and Galiani, 2015; Baranzini *et al.*, 2017).

By making fossil-fuel energy and carbon-emitting technologies more expensive, an economic instrument encourages conservation, and promotes the use of low-carbon energy sources as a substitute. As a result, it drives more environmentally efficient production and supports the case for investment in abatement technologies and research and development (R&D), ultimately lowering the costs of achieving the climate goals (The Royal Society, 2002; Moarif and Rastogi, 2012; UNCTAD, 2016).

This positive relationship between high energy prices and the development of less carbon-emitting technologies is evidenced in Ambec *et al.* (2013), where Jaffe and Stavins (1995) empirically find, that in the presence of stable carbon prices, innovations tend to reduce consumer prices and make equipment more energy-efficient when oil prices are high. A high carbon price signalises for the near and more distant future, the need to reduce dependence on high carbon energy in all economic sectors via innovation (Baranzini *et al.*, 2017).

This demonstrates that carbon pricing is an important element of a policy mix aimed at fostering a transition to a low-carbon economy. For society, there are large co-benefits in reducing energy costs such as improved energy security, increased efficiency, less dependence on emissions-intensive sources within a more diverse fuel mix, besides preservation of the local environment and an improvement in local air quality. This in turn limits the risks to human health in the mid to long-term (PMR and ICAP, 2016; Bollen et al., 2009).

One useful characteristic of pricing pollution to control emissions is that it raises public revenues to support public priorities, which can reduce the policy costs as well as increase the acceptance of carbon pricing. These revenues could be usefully applied to induce a wider adoption of low-carbon technologies by making the final price of this energy artificially less expensive, to promote mitigation in other sectors, or for other policy purposes such as revenue recycling, as a support to economic sectors that are adversely affected. The main problem with subsidising emissions-intensive activities is that the economic support given may encourage overproduction of output and over-emission levels, undermining the climate policy and accelerating environmental effects. According to many studies, these unmitigated effects would unequally affect countries, with more pronounced negative effects on the poorest (Tol *et al.*, 2004).

Another possibility is to utilise those revenues, at least partially, to compensate the regressive effects of increased costs on poorer households, for example, via tax reductions for low income or energy poor households, or lower value-added tax (VAT) rates for products serving basic needs (Bowen, 2015).

Overall, an ETS market system is expected to generate a more unequal distribution of wealth, which can be avoided by an ETS design that delivers a just outcome, especially through the definition of rules for allocating permits (Caney and Hepburn, 2011).

The simplest, and from an administrative perspective, less burdensome manner of recycling revenue from carbon pricing, is to implement a lump sum redistribution (Metcalf, 2009). Lump sum is referred to a distribution mechanism where every consumer equally receives revenues. In this case, relative choices among goods or relative profitability of economic activities are not affected. Even though revenue recycling is one relevant instrument to avoid undesirable distributional impacts, it requires an appropriate policy design (or other complementary measures).

Therefore, it is important to structure market-based transfers properly in order to account for distributive justice and wealth distribution between regulated sectors (and countries) as well as to ensure the environmental outcomes. A well-functioning set of political institutions is fundamental, not only to alleviate any distributional impact, but also to help the market to work more effectively. A strong political institution also assigns penalties on violators and supports monitoring of emissions and other technical requirements for successful implementation, whilst aiming to prevent fraud and corruption.

In this context, getting the most appropriate policy design, within a reasonable and stable price structure, as well as using the potential revenues in productive ways to provide the correct stimulus on investments, plays an important role in climate governance. Ultimately, these aspects in combination shape the character of the market, and drive necessary changes (Commission on the Economy, 2014).

However, carbon pricing is one among several instruments to tackle a wide-range of market failures in both developed and developing countries. At the required time, it may not be sufficient, as well as being too risky as a single instrument (Edenhofer *et al.*, 2009; Stavins *et al.*, 2014; Baranzini *et al.*, 2017). Therefore, efforts should focus on creating in parallel, additional mechanisms such as modifying the composition of demand (towards less energy usage), promoting structural change in the composition of the economy (from the predominance of emissions-intensive sectors and products and inputs to a lower-carbon production), as well as promoting low-carbon transportation and energy sources.

For that purpose, a key objective is to combine a broad range of areas within a fiscal reform in a way that enhances the performance of each of the policies, addressing competitiveness, trade and investment, labour market, human capital and education, innovation, among other issues (PMR and ICAP, 2016; Rydge, 2015; Commission on the Economy, 2014). The above list of potential complementary policies is not exhaustive, and does not necessarily imply a price-based control of emissions from sources other than energy production and use, such as landfill emissions or forestry.

Altogether, these strategies would change behaviour and support conditions for decarbonising the economy whilst allowing for development, which has an acceptable implication for society. Additionally, they could reduce the estimated economic impacts of mitigation (Jorgenson *et al.*, 2008). Under this framework, the use of market-based instruments is a consistent strategy to manage change beyond the creation of carbon markets to broader forms of structural transformation (Newell and Paterson, 2013). Hence, better performance is likely to occur in countries which anticipate and plan for change.

When there is no clear communication on the strategies proposed to aid a low-carbon economy transition, objections may emerge. The major source of resistance of governments to climate policies so far has been the potential costs of reducing emissions, and the respective negative effects (Harrod and Martin, 2014). On one hand, the environmental policy and goals tend to be consistent with broadly held societal values, but eventually the targets are met at excessive costs, at the expense of the economy or the environment. This can occur as a result of the mitigation policy being overly stringent or too lax, respectively (Harrington, Morgenstern and Sterner, 2004; Commission on the Economy, 2014). On the other hand, there is an argument for opposition on the "moral" grounds of commodifying the atmosphere (the marketability of environment), which grants regulated entities permission to pollute subject to a fee, and in some conditions allows the alienation of mitigation responsibilities (Lohmann *et al.*, 2006; Page, 2011).

Other concerns are mostly justified by negative side-effects on energy prices, transaction costs and employment creation, among other potential sources of pressure appointed by those who might somehow be affected, i.e. industries, government, organisations and individuals. There appear also to be a lack of political will and technical know-how.

Most importantly, there is fear of facing competitiveness[15] loss, particularly due to free riding and emissions leakage, a major issue for the effectiveness of a climate policy. In theory, emissions leakage originates from differences in environmental regulations between countries and implies an increase in emissions in one jurisdiction as a consequence of emissions reductions in another. Hence, there are incentives for polluting activities to shift away from jurisdictions where engaging in mitigation is mandatory to jurisdictions where regulation of the climate is less stringent, since production costs are lower. This is commonly referred to as carbon leakage.

Driven by the relative costs of stringent regulation, this reallocation of production has mainly been in a one-way direction, from developed to developing countries. The main effect has been a shift in comparative advantages and trade patterns, for example increasing emissions, and leading developing countries to a specialisation in emissions-intensive sectors, as defined by the pollution heavens hypotheses[16] (Cole, 2004). This motivates the need to coordinate actions between countries and emitters in order to prevent free riding and leakage at international level. With an international climate agreement, the implementation of a global carbon price could guarantee the absence of carbon leakage and free riding, since it levels the playing field (Baranzini *et al.*, 2017).

This difference between economic theory and political reality explains the reason why economic instruments have not prevailed in environmental policy

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[15] See Dechezleprêtre and Sato (2014), Taschini (2009), UNEP (2004) and Stavins (1997).
[16] There are two main mechanisms to address this issue: to impose carbon-related border adjustments or, instead, to emphasise mitigation under the perspective of consumption of goods and services, rather than production.

packages over the world. However, given the slow progress in addressing climate change through traditional regulation, the resistance to adopting economic instruments by policymakers is slowly fading away. The growing popularity indicates the inevitable need to incorporate them in the policy mix as a key element to a sustainable energy policy in the response to climate change (The Royal Society, 2002; Harrington, Morgenstern and Sterner, 2004; Barrett, Carraro and Melo, 2015). Meanwhile, some authors such as Seroa da Motta (2006) suggest a gradual introduction of the market-based instrument in order to develop the necessary structure, legal, and institutional capabilities to support carbon pricing and thereafter obtain better efficiency of the environmental policy.

Nevertheless, the decision on the appropriate instruments for the policy mix to address climate change is a complex and challenging task, not only because climate change features must be accounted for in the policy formulation to ensure cost-effectiveness, but also because they are developed based on specific political and economic contexts within a particular market structure. With regard to energy use, which is closely linked to economic structure and the level of development, this is particularly relevant (Moarif and Rastogi, 2012).

Furthermore, the instruments vary in terms of total costs, distribution of costs among emitters and other segments of society. Changes in emissions' trends or potential costs and benefits of abatement of climate changes in the long run cannot be predicted precisely. For policymakers, this implies a degree of foresight and willingness to act without getting a rapid response.

Typically, the best policy instrument is the one capable of meeting the target with a high level of reliability (Perman *et al.*, 2011). However, there are multiple objectives in the policy arena, and the instrument choice usually reflects that. Accordingly, it will depend on relative weights attached to the criteria it aims to attain and the existing political, economic and environmental priorities. For example, national policies to help achieve higher development level or to reduce poverty have been assigned a primary relevance over environmental problems in developing countries, which is reflected in the late incorporation of climate issues into their domestic agenda.

To design appropriate climate policies for national circumstances, the level of development is particularly important. Besides the influence it exerts on the capacity to mitigate and to promote sustainable development, it also determines the best policy option to balance costs with mitigation associated benefits, whilst concurrently enabling human development (Clarke *et al.*, 2016). By envisioning carbon pricing mechanisms, developing countries perceive this strategy as attractive due to the financial opportunities involved.

Yet, note that overall performance of the selected method is subject to factors such as macroeconomic and energy profiles, development levels, domestic and international pressure (through international climate forums). There is evidence that lobbying by energy-intensive industries aimed at receiving a more favourable treatment has resulted in less effective carbon pricing policies (Demailly and Quirion, 2008; Baranzini *et al.*, 2017).

Literature shows that, over the last two decades, experiences with the existing instruments have provided different outcomes and lessons to address these issues. From these insights, policy proposals based on economic incentives have emerged, being increasingly more extensively accepted. Mostly in developed countries, this option to mitigate emissions associated with economic production is becoming a prominent option in developing countries, particularly carbon pricing through cap-and-trade schemes. This proposal is quantitatively evaluated in this study for Brazil. The next topic describes briefly the functioning of carbon pricing in its different forms, and focuses on exploring the design elements of these schemes and other relevant aspects.

2.2.2 Carbon pricing: design elements of Emissions Trading Schemes (ETS)

There are broadly three market-based mechanisms underlying efforts to decarbonise the economy: subsidies, taxes or transferable permit markets, as described in Table 1[17]. Among them, the most commonly used are carbon pricing instruments, namely emissions taxes (typically carbon) and the so-called ETS. Even though both are good approaches to the climate problem (Stavins, 2008), there are key similarities and differences that make an ETS a preferable carbon pricing instrument choice, although there are several carbon taxes cases over the world. Advantages and disadvantages of each approach depend on the system's design, but literature highlights greater opportunities for opting for an ETS.

Subsidies for pollution abatement are financial assistance generally adopted to complement direct regulation, as a support to meeting the environmental standard, for instance. They can take the form of grants, subsidised loans or fiscal incentives, or instead, removal of harmful subsidies. A good example is phasing out subsidies for the production and consumption of fossil fuels, particularly if part of a broader fiscal and energy sector reform (Rydge, 2015). For Goulder and Parry (2008), subsidies might be compared to the other two emission pricing policies[18] since every additional unit of emission implies a cost to the firm in forgone subsidy receipts.

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[17] There is also the deposit-fund system, which is a combination of a tax on product consumption with a rebate (subsidy) applied when the product or its package returns for recycling. There are experiences with this mechanism in USA, Germany, Sweden and Norway.

[18] Although, when comparing profitability the effect of a subsidy per unit of emission reduction should establish the same abatement activity as a tax of the same magnitude per unit of pollution emitted. Taschini (2009) differentiate the former from its increase profits and the latter for its reduction in profits.

Table 1 – Categories and description of main market-based instruments

Type		Description
Subsidies	Grants	Monetary assistance for the implementation of anti-pollution measures
Subsidies	Subsidised loans	Financing of anti-pollution investments at an interest rate below market rates
Subsidies	Fiscal incentives	Tax rebates or other forms of exemption in cases of adhering to anti-pollution measures. Removal of subsidies that stimulates consumption of carbon-based fuels or subsidies which encourage investments in low-emissions energy sources and energy efficiency
Taxes	On effluents	Payments for discharges into the environment based on quantity and/or quality of the effluent. Example: carbon tax
Taxes	On users	Payments for the costs of public treatment or effluent collection, which can be charged evenly or differentiated in accordance with the quantity of the treated effluent. Example: tariffs for water or sewage treatment
Taxes	On products	Additional costs imposed in the final price of products where production is pollution-intensive. This is an indirect emission pricing mechanism that generates additional revenues for the government. Example: fuel taxation
Markets	Negotiable emissions permits	Purchase and sale of pollution rights specified in quotas, which can be distributed among regulated sectors and industries
Markets	Mandatory environmental insurance	Transfer of liability for environmental damages made by polluters to insurance companies
Markets	Market support	Government intervention via price to stimulate the market for secondary material such as recyclables

Source: Elaborated from Rathmann (2012), Harrington, Morgenstern and Sterner (2004), Costanza *et al.* (2014) and UNCTAD (2016).

Within the same broad category there are also regulations that are designed specifically for flexibility, the carbon pricing mechanisms. The major difference distinguishing the carbon tax and an ETS lies in the mechanism of imposing the limit on emissions, as the former directly places the price on a tonne of carbon, and leaves to market forces any resultant reduction in fossil fuels and emissions, whilst the latter relies on the market to establish the appropriate price for carbon that generates emissions reductions based on the specified target. Accordingly, higher carbon prices are only possible if raising the tax rate of a carbon tax system, or if permit availability is reduced in the ETS system.

A carbon tax (also known as “Pigouvian tax”[19]) is set on a price basis on effluents, user or products in order to equalise the marginal social damage. However, it controls the carbon-based energy at the specified tax level without guaranteeing a certain level of reduction, since the quantity of emissions fluctuate according to variations in the demand for energy. Although this provides great flexibility to regulated sectors to be certain about costs and determine the lowest-abatement solution for mitigation, a carbon tax system is only environmentally effective if real market incentives are created, which depends in a large part on whether the tax rate is high enough.

At the same time, a tax on emissions raises the cost of production if it is levied at the point where fossil fuel passes from the producer to the next entity in the supply chain, and raises the product price when passed along in the rates consumers are charged (Revelle, 2009). As a result, it also affects the demand thereby inducing a change towards more energy-efficient or low-carbon products. From the producer perspective, this approach encourages the search for ways of economising on energy use and switching to lower-carbon sources and technologies.

In the long-term, an economy-wide tax with a planned escalation of price over time indicates stringency, being able to provide a consistent price signal for sustaining climate-friendly investments. In fact, the price predictability is an

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[19] The term is a reference for Arthur Pigou, who proposed a corrective tax on pollution to balance the social cost and private cost of externalities.

important advantage of carbon taxes. Another very attractive feature is the capacity for directly raising substantial revenues, as noted in the previous section.

This particular climate policy can be very regressive since a large part of lower-income households' income is spent on energy-using goods and services, with a low degree of flexibility to adjust. In the literature, there is evidence for this in Ireland in Verde and Tol (2009), in the United Kingdom (UK) in Feng *et al.* (2010), in France in Bureau (2011)(Bureau, 2011), in the United States (USA) in Rausch, Metcalf and Reilly (2011), in India in Ojha, (2011) and for Brazil in Magalhães (2013). Using revenues to reduce this regressive effect, to rebate equally to individuals or as rewards for those saving energy from limiting the consumption could, accordingly, neutralise the related impacts.

In this sense, deciding on how revenue is allocated is very strategic, as well as deciding on the appropriate price. Further considerations in terms of the carbon tax system design include choosing the emissions coverage, the point of taxation (upstream or downstream), and the flexibility of the price to change as a result of new information on the marginal cost of abatement (Narassimhan *et al.*, 2017). In practice, carbon taxes face greater political opposition in light of the well-known administrative structure of taxes, which could lead to exemption of sectors and firms and as a result, reduce environmental and cost-effectiveness (Stavins, 2008; CRP, 2017).

Generally, carbon taxes are simpler since it does not involve capacity building to manage the system, and are more transparent compared to ETSs but overall benefits and costs to society are very similar[20]. Both mechanisms imply an upward effect on prices and costs throughout the economy with a system designed to help meet climate goals at minimum costs, and that encourages the switch to non-fossil fuel energy sources, whilst penalising emitters. Yet, country-level experiences of carbon taxes remain fairly limited, as opposed to the progress ETS has had in recent years (Lanzi *et al.*, 2013).

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[20] These papers offer a good discussion in this regard: Andersen and Ekins (2009), Taschini (2009), Freebairn (2010) and Goulder and Schein (2013).

An ETS has an appealing virtue and garners political support from a wide spectrum of relevant actors, as there is the capacity to promote significant emissions reductions (Taschini, 2009; Caney and Hepburn, 2011). As ETS allows for more stringent targets to be adopted, which consistently get tighter over time. This secures the greatest environmental effectiveness (and environmental integrity) whilst progressively increasing the costs of pollution. In theory, the carbon price assures that the level of emissions do not exceed the limit, and if coupled with a robust Monitoring, Reporting and Verification (MRV) system for compliance, the climate commitment can be met in the long-term. In addition, it appropriately maintains the costs of compliance through trading, and fosters competitiveness in low-carbon intensive technologies.

An ETS system, often referred to as "cap and trade"[21], is a quantity-based[22] economic approach where the maximum amount of emissions is determined by setting a cap, and thereby translated into a number of tradable permits in the market which is issued to regulated emitters, whether installations, firms or sectors. These property rights[23] on emissions can be distributed at different jurisdiction levels (subnational, national and international), depending on the determined coverage. Whilst the binding limit creates scarcity of allowances and the price incentive, trading enables the emitter to adjust to the target in the most cost-effective way. In fact, the degree of heterogeneity of costs and abatement opportunities across regulated entities is one major advantage of an ETS. Thus, the inclusion of as many sources and participants as possible is necessary to provide greater benefits from trading.

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[21] Emission trading is a broad concept and includes not just cap-and-trade schemes but also rate-based trading and project-based mechanisms that are often linked to one of the former regimes. See Philibert and Reinaud (2004).

[22] This characteristic distinguishes an ETS and carbon prices, and approximates its functioning to command-and-control instruments. However, an ETS allows the transferability of permits, or the "right to pollute", between regulated sectors thereby creating opportunity costs. For Perman *et al.* (2011), this suggests that ETS, tax or subsidy systems work in equivalent ways.

[23] This assignment of property rights was firstly proposed by Coase (1960). The idea is that environmental problems can be addressed as long as property rights on the use of the damaged resource (or pollution) are defined. As a result, economic actors are free to negotiate, leading to an optimal level of externalities if there is no transaction costs involved. This has become known as the theorem of Coase.

Consequently, this liquidity reduces the risk of market power since allowance availability is not subject to the control of certain economic actors.

In an ETS, each participant is granted emissions rights for every tonne of emissions for which they are accountable and, according to their performance in limiting emissions, can buy or sell permits in the open market. In other words, the permits become an input to production, and can be traded accordingly. Each addition of a unit of emission implies a cost equal to the allowance price, and thereby the allowance cannot be exchanged. For those firms unable to easily curb emissions at low cost, the alternative is to purchase additional permits from emitters who find it relatively cheaper to undertake extra abatement, to invest in new technologies for mitigation, or employ measures to reduce related costs. These organisations benefit from selling unused allowances at a market price.

This price expresses the marginal cost of emissions reduction, being determined endogenously by market forces, that is, the balance between supply and demand along with other factors related to the market structure (Taschini, Fankhauser and Hepburn, 2011; Chevallier, 2012). As long as the marginal cost of abatement is below the price, there is stimulus to further reduce emissions and sell permits. This is particularly the case with a more stringent cap, as it results in a lower supply of allowances, consequently higher prices and a stronger signal towards mitigation. Hence, regulated entities may judge that the best strategy is to comply with the emissions target, otherwise a punitive fine is applied by the regulator, which has the responsibility to surrender allowances and guarantee enforcement. For those covered by the ETS, the costs of doing so are lower than without trade and the corresponding trading profile reflects the emitter's structure of abatement costs (Manzoni, 2013). Nevertheless, it also depends on how the ETS is designed.

Within the ETS structure, there are several different design features available to enable a successful climate policy in terms of cost-effectiveness and environmental integrity. For each variant, a series of decisions and actions are undertaken, in a combination that will shape the functioning of the market. At first, developing a system requires clear identification of the policy objectives in order to include the most appropriate elements, and gauge the intensity in which they should operate in the market. Some factors should be taken into consideration in this regard, such as the contributions of an ETS to emissions reductions targeted, the acceptable level of costs, distribution of costs and benefits, generation of revenues and how to use them, or the ETS could contribute to economic transformation and sustainable development (PMR and ICAP, 2016).

Once the objectives have been established, the ETS design options can be settled in. This involves deciding on the scope, cap, method of allowance allocation, adoption or not of temporal flexibility arrangements and other price/cost containment measures, rules for new entrants, the best way to ensure compliance and oversight in the defined period of compliance, and finally, to consider a potential cross-border linkage. In general, there are some criteria to choose among the different approaches in every element, whether the effective contribution to mitigation, system predictability and flexibility, accountability and transparency or the appropriateness to local objectives and contexts as fairly as possible. From those, the precise ETS design is tailored to the jurisdiction in order for it to function effectively, thereby prompting a broad public acceptance and giving confidence on the system endurance.

Among these elements, perhaps the most critical is the scope and the cap set, which implicitly determines the price in the trading market. The ETS's scope refers to sectors/installations of the economy and types of GHGs covered under the system. Sectors differ according to the composition and size of the economy, as well as the emissions and abatement opportunity profile. From this perspective, the larger the number of emissions sources covered, the better it might be, particularly if the proportion of emissions these sectors account for is significant (Flachsland, Marschinski and Edenhofer, 2009b; Manzoni, 2013; Mehling and Görlach, 2016). The main cost-effective benefits are encountered where abatement costs across participants are heterogeneous (Rezek and Blair, 2005; Newell and Stavins, 2003; Sterner, 2002; Tietenberg, 1985), which lower compliance costs, address carbon leakage and competitiveness effects on regulated sectors, and allow a better functioning of the permit market.

However, covering sectors composed of small firms with a limited share of emissions is likely to increase administrative costs, while achieving lower mitigation outcomes. One approach is to introduce a threshold so to protect small firms that, in turn, are not subject to the ETS requirements. Another motive for locking some sectors out of an ETS is the inability to measure or monitor emissions with sufficient certainty. Without appropriate mechanisms for monitoring and enforcement, there are not enough incentives for a higher degree of compliance since it is rather difficult to match emissions and permits (Coria and Sterner, 2008). However, this also depends on the technical ability to ensure a strict enforcement by detecting violations and penalties.

In most cases, the coverage of an ETS is comprised of a reduced number of sectors with large emitters of CO_2, where measurement is more accurate, typically the power and energy-intensive sectors. The costs of monitoring these emissions depend on the point where they are regulated. The point of regulation plays an important role since it influences behaviour change, whether by regulating entities responsible for processing and commercialising emissions sources (usually a fossil fuel) - upstream, or at the point of supply where these are physically released into the atmosphere - downstream. The latter is considered to be a preferable option in light of the existence of downstream data and compliance mechanisms.

As for the cap setting, the desirable level of emissions reductions within the covered sectors is specified for the compliance period to reflect the environmental objectives. This is an important element for policymakers to balance the commitment-flexibility trade-off in an ETS (Acworth *et al.*, 2017). In addition, the cap tends to be aligned with the economy-wide emissions reduction target, in a way that the mitigation responsibility is shared among ETS and non-ETS sectors. This influences the share of emissions reductions within the ETS system as well as the level of ambition.

The ETS cap is a determinant of committing to mitigation. The degree of ambition typically takes into consideration the existing trade-offs in terms of mitigation and costs to capped (and non-capped) entities as well as the timeframe to achieve proposed emission reduction targets. The logic is that the more ambitious, the lower the number of allowances and the higher the costs faced by those under the cap.

Two important cap features can affect the way investments are planned, the cap period and the baseline against which emissions are to be reduced, namely historical (base year) or projected future emissions (i.e. against a business-as-usual –BAU– scenario). An ETS usually operates by assigning cap periods, which are adjusted linearly or exponentially downwards to create stringency and to enable adjustments in behaviour of regulated entities to comply. Maintaining the cap consistently over time is also important to guarantee both carbon predictability and continuous abatement (Zeng, Weishaar and Couwenberg, 2016).

In most cases, the cap is set at an annual or multiple-years basis, in coordination with a commitment period, for instance, international pledges of mitigation or ETS phases under which other ETS specifications are determined. A short-term cap allows the regulator to adjust the targets to marginal abatement costs and technological progress. On the other hand, a long-term cap may provide signals on the cap trajectory, especially if updates are projected, so to allow regulated entities to realise low-carbon investments and plan temporal abatement. Additional certainty on abatement costs and price stability can be assured through flexibility mechanisms (Fankhauser and Hepburn, 2010), as will be discussed later.

At this stage, the infrastructure is established and the participants are familiar with the ETS functioning. Based on that, PMR and ICAP (2016) starting with a less stringent cap and gradually adjusting it. In a recent study, Narassimhan *et al.* (2017) have identified several design features to allow cost-effective emission reductions and price stability in the initial phase of an ETS. One of them is rolling out dynamically-adjustable emission caps. This supports the idea that the cap can be scheduled to accommodate new emissions data over time or to introduce other ETS features, such as mechanisms for allocating allowances, adding new sectors or initiating an international cooperation through linking. Therefore, the cap setting should be made iteratively with other ETS design elements. If perceived as credible and fair, the ETS design gives rise (or maintain) to political acceptability (PMR and ICAP, 2016), although negotiation is very complex (GVces, 2013; Moore, 2013).

There are two methods of capping emissions. Both are able to deliver ambitious mitigation outcomes if the target is set up to reflect that, but at different costs. The first one defines the maximum absolute quantity of emissions allowed and finds support in being simple and objective in fixing the adequate volume of emissions, but also because it helps price predictability and credibility, a condition for forming expectations and making decisions at the regulated entity level. However, “absolute caps are often viewed as a constraint both to economic growth and to the right to (sustainable) development” (Hamdi-Cherif, Guivarch and Quirion, 2011, p. 4). Hence, the cap specified does not consider potential up or down deviations. If it is above expected, then there will be more mitigation, hence a higher cost. To address concerns over the uncertainties of economic performance, an alternative has been proposed.

The second is an intensity cap, which determines the limit of emissions according to the indicator in which it is indexed. A relative target is expressed as an emissions rate per unit of output or activity, mostly measured by Gross Domestic Product (GDP) or energy consumption. With this cap, emissions variations follow the performance of the reference variable. For example, if based on GDP, and this is higher than projected, it will allow emissions to rise[24]. In other words, the cap is not fixed and may be ex-post adjusted when economic activity deviates from the projected level, possibly relaxing the limit of emissions (Zeng, Weishaar and Couwenberg, 2016).

The major argument against this approach is the uncertainty of the environmental outcome and problems of implementation, mainly associated to administration and monitoring (Baron and Bygrave, 2002; Dudek and Golub, 2003; Müller and Müller-Fürstenberger, 2003). Compared to an absolute cap, an intensity-based cap would serve as an output subsidy to covered entities and

[24] The relative cap, or the quota for the year t weighted by the variation of output, can be calculated as follows: $RCAP_t = CAP_t X \frac{Y_{t-1}}{Y_{t-2}}$, where CAP_t is the limit of emissions in year t, which is equivalent to an absolute limit, and Y_{t-1} is production of previous year and Y_{t-2} is the production of the two previous years. More information on that is available at Manzoni (2013).

undermine the climate policy (Gielen, Koutstaal and Vollebergh, 2002; Fisher, 2003). Moreover, this approach could fail to provide the necessary stimulus to change the demand for emission-intensive sectors.

One the other hand, it may be perceived as preferable for economic sectors due to cost certainties (Rusche, 2010). Wing *et al.* (2006) indicate that it could be more appropriate to developing countries by reducing the uncertainty of economic outcome from mitigation, which could be consequently lower, and to increase willingness to participate in international agreements. For developed economies, where decarbonisation within few decades is expected to be accompanied by a moderate potential for growth, an absolute-cap structure is usually indicated (PMR and ICAP, 2016).

Note that decisions on the cap structure have implications on the allocation of permits, as those are issued to participants according to the number of permits available under the cap. There are three methods that orientate the distribution of emission permits in an ETS. In the first case, the regulator allocates them at no cost. The number of allowances each emitter receives can be determined proportionally by their historical level of emissions, which is referred to as grandfathering, or ex-ante allocation.

The attractiveness of such an approach has been associated to its capacity to compensate the sector-specific costs of purchasing emission permits on the market (Ellerman, Jacoby and Prinn, 2009) and to correct competitiveness distortions (Branger *et al.*, 2014), which is important for political acceptability. Some regulated sectors are able to pass through carbon-related costs, but for those competing in the global market it may be difficult, which encourages the reallocation of economic activity and its associated emissions to other regions and risks the integrity of the policy (Zetterberg *et al.*, 2012).

In order to protect against these limitations, the other method of free distribution sets the quantity to be allocated based on benchmarking, that is, according to an industry baseline, whether calculated at a specific emissions rate or on the basis of share of output (an indicator of efficiency). Sources emitting below the benchmark are rewarded with a proportional quantity of allowances, often greater than those who emit more. This effect is problematic also with grandfathering because it could yield a reward for large emitters, thereby discouraging early action while expecting to acquire more emissions in the future (Manzoni, 2013).

In both cases, there are distorted incentives to undertake cost-effective abatement, leading to extra gains (so-called “windfall profits”), and no revenue is generated. Keohane, Revesz and Stavins (1998) argue that choosing grandfathering in most systems is a direct consequence of political influence regulated firms play in the policy process. In practice, grandfathering of incumbent emitters is a way to bring about high-emission sectors and countries by accommodating self-interest and gain political consensus (Posner and Weisbach, 2010), the reason why it has been applied virtually in all applications to date (Coria and Sterner, 2008).

According to Rathmann (2012) and Jegou and Rubini (2011), an ETS rolled out with free distribution seems to be important to enhance political acceptability in initial phases, thereby easing the transition and gradually introducing the carbon costs through auctioning. If combined with ambitious targets, this could simultaneously generate revenues over time (Narassimhan *et al.*, 2017). The recommendation of PMR and ICAP (2016) is that:

> The decisions on the trade-offs between ambition and cost may change over time. In the early stages of an ETS, the government may place a higher priority on getting the fundamental ETS architecture in place, building support for the system, and getting started with trading, rather than achieving an ambitious level of mitigation at potentially high cost. Applying a relatively higher and, thus, less stringent cap in earlier periods can also help lower the perceived initial risks to participants and to the economy; reduce competitiveness impacts; and create an enabling framework for the necessary learning processes for regulators, regulated entities, and other stakeholders (PMR and ICAP, 2016, p. 47).

From the economic perspective, auctioning appears to be preferable due to higher opportunity costs of trading[25] in the market, and at the regulator’s point of view, because it generates public revenue for augmenting the system’s

[25] Within an ETS, efforts to mitigate and sell allowances in the market configure a traditional example of rent-seeking behaviour (Jegou and Rubini, 2011).

efficiency or to be used in other programmes. It is an administratively simple and efficient method to allocate permits, through a mechanism of auction, to those who value them the most and accept to pay the emission price. In some circumstances, only a certain share is auctioned at a rate that could change over time. Besides rewarding regulated entities that undertake measures to abate, it also incentivises early action. Once the ETS is implemented, there is also lower opportunity for lobbying, in favour of a certain sector, although some may still exist for the auction process. Compared to other forms of distribution, auctioning offers the most straightforward and transparent method of allocation, with reduced risk of distortions (Gilbert *et al.*, 2006).

The timing and volume of auctions are often announced in advance, to avoid disturbances. In fact, because participants have to pay the full cost of allowances, windfall profits are reduced and help ensure the functioning of the trading market. Because of these same costs, the disadvantage pointed out against auctioning is the lack of direct protection against emissions leakage as well as compensation for losses, particularly for those sectors exposed to international competition. From the political perspective, this could cause barriers to the ETS adoption. Other related-concerns relate to how small firms can participate in the auction process without raising further transaction costs, which can lower the ETS effectiveness (Coria and Sterner, 2008).

However, deciding on how to distribute allowances is critical because it defines the distributional impact of the ETS system, thereby determining how effective it could be in addressing competitiveness and carbon leakage issues. The climate policy implies a redistributive effect between covered sectors, and there will be beneficiaries and losers. At the individual's level, it may involve shifts in income, which means it alters the distributive justice.

The question posed is what allocation arrangement of an ETS aimed at mitigation, is economically efficient and fair. Caney and Hepburn (2011) affirm that an ETS can be designed to deliver a just distribution, either through gratis allocations of allowances, or by auctioning and distributing revenues, which could be rather progressive. It is known that markets are incomplete, where production and consumption behaviour generate uncompensated external effect upon other. As noted earlier, these adverse effects on economic actors can be managed subject to additional inclusion of compensatory measures suitable to all involved. For developing countries planning ETS systems, distributional impacts can be rather detrimental.

To facilitate compliance and reduce distributive implications, there are some additional flexibility provisions to build into the ETS design, which allows emitters to shift reductions to a lower-cost time period through banking or borrowing. In other words, to decide when and where to undertake emissions reductions between now and some point in the future. In addition, it may help coping with uncertainties such as production levels (Sterk and Mersmann, 2011). This temporal flexibility has been recognised as fundamental to self-correct the dynamic of the market since it controls price volatility and minimises the related costs throughout the policy duration (Eden *et al.*, 2016; Acworth *et al.*, 2017). In this sense, it diminishes the exposure to market risks and transaction costs (Manzoni, 2013).

Banking allowances has the advantage of enabling shifts in industry structure by phasing out old technology at a time that is cheapest to do so, and promote further mitigation now if a tightened cap is expected in the future. Further, it can hedge against unexpected price surprises (Jorgenson *et al.*, 2008). Environmentally, the main limitation of banking is the difficulty in correcting over-allocation of an ETS in a later trading period; otherwise, banked allowances indicate that more emissions have been reduced than the required level, at least in initial years. On the other hand, borrowing emissions from future budgets is neither environmentally effective nor sustainable as it postpones mitigation to the future, with the risks of no mitigation at all if regulated entities rely on borrowing to artificially increase future compliance costs to claim for softer targets (Sterk and Mersmann, 2011). Failure to implement a well-design choice for either banking or borrowing leads to price collapses from one period of functioning to the next (Schmalensee and Stavins, 2015).

In general, allowance prices fluctuate as a result of a balance between supply (mostly controlled by the regulator) and demand (driven by a complex iteration of the economic and organisational-level factors, which indicate that price signals about abatement costs are transmitted to regulated entities (PMR and ICAP, 2016). However, some factors such as exogenous shock, regulatory uncertainty and market imperfections can result in excessive or persistent price variability, which is detrimental to decision-making about investments and justifies the need to intervene in the market.

In addition to temporal flexibility, another feature embedded in the overall market design to address uncertainties of mitigation costs is to establish upper and lower bounds for allowances price, also known in the literature as a “safety valve” (Pizer, 1999). They are cost-management provisions, whereby the cost of capping emissions at the specified target level is limited to a safety price valve. If economic growth or other factors were to cause a variation in the permit price differently than expected, the mechanism impedes the price from failing below or increasing above a pre-set price, corresponding to a price floor and price ceiling, respectively.

In practice, there are different ways of enacting a floor price, which limits the amount of permits available in the market. For instance, by keeping allowances out of circulation unless the purchasers pay the minimum price or whenever the floor price is reached to avoid any further price reduction. When permits are auctioned, it involves setting a reserve price (Hepburn *et al.*, 2006). Inversely, the ceiling price guarantees that additional allowances are returned to the market whenever the price exceeds acceptable levels, i.e. when the pre-set ceiling price is reached (Goulder and Schein, 2013). Such a reserve is known as a cost containment reserve (or market stability reserve) and can be deployed every time the price crosses the determined threshold.

Whilst low prices can yield windfall profits, a price perceived as excessive could undermine the viability of the ETS. In either way (or in the combination of them, the so-called price collar), the use would serve to redirect the form of controlling emissions, emphasising prices, rather than quantities (Jacoby and Ellerman, 2002). Therefore, their usefulness will depend on the objectives of the climate policy and whether benefits exceed the risks (PMR and ICAP, 2016). If the objective is to put a price on carbon and achieve a balance between abatement costs and emissions reductions, a much lower (or higher) carbon price may amount to policy failure (Wood and Jotzo, 2011). At the same time, evidence from the literature suggests that is rather more beneficial to use these arrangements than adopting a pure ETS approach (Pizer, 2002; Burtraw *et al.*, 2013).

There is another option to keep the permit price from rising and thereby reduce the compliance costs - the offset. This is a popular way to introduce a hybrid system (Grüll and Taschini, 2011). By allowing offsets, the ETS warrants outsourcing of required emissions reductions elsewhere in order to count against the requirements. This is based on the notion that location of mitigation plays a secondary role. Offset projects[26], predominantly financed in developing countries, give capped sources access to other abatement opportunities cost-effectively (Newell and Stavins, 2003; Jorgenson *et al.*, 2008). Hence, mitigation from these projects generates credits equivalent to permits to help comply with the overall cap, subject to a limit of usage. Making adjustments in this share of offset can further smooth price fluctuations (so-called offset relaxation). A critique of this mechanism is that instead of constraining domestic emissions, it pays for others to have the responsibility of lowering emissions (Page, 2011).

Once the phases are determined, rules to accommodate the abovementioned provisions in the ETS design can evolve across periods of compliance to seize the best cost-effective design. This means taking into consideration a broader sectoral and emissions coverage domestically, or instead, across borders through the harmonisation of ETS strategies at international level. For Stavins (2008), enabling such alignment of climate policies in a well-designed fashion, that is politically feasible, may favour the adoption of an ETS. Even though an ETS does not entail environmental protection, but ultimately the right to use a certain proportion of the absorptive capacity of the atmosphere for a period of time, it is simply a means to an end (Caney and Hepburn, 2011).

[26] There are different categories of projects suitable for generating offsets credits, for example, those for renewable energy, methane abatement, energy efficiency, reforestation, fuel substitution, etc. However, the credits are only assigned to projects where mitigation would not have happened otherwise (a case of additionality).

Obviously, it does not operate in isolation. Recently, research has found that in the presence of cost uncertainties, a hybrid system with a tailored combination of ETS and taxes has efficiency advantages and could engage a wider set of sectors and countries in combating the climate problem (Schmalensee and Stavins, 2015). According to Jaffe, Ranson and Stavins, (2009) and Jotzo and Betz (2009), the largest economic benefit of a cap-and-trade scheme stems from the integration to other schemes. Indeed, international cooperation through carbon pricing has gained increasing relevance in the climate policy architecture since national policies have shown to be insufficient to promote mitigation of climate impacts (Stavins *et al.*, 2014). Given the potential of a global ETS in the future, the benefits of early participation are significant (Flachsland, Marschinski and Edenhofer, 2009b).

Similarly, to ensure that a formal linked ETS system will deliver the environmental and economic outcome envisaged by policymakers, it has to be designed accordingly. The linkage design results from the combination and interplay of the features of participating ETS systems, with differences reflecting domestic political consensus as well as economic and environmental priorities. Negotiations are very challenging and therefore, imply reconciling sufficient common elements.

Although the literature supports that linking does not necessarily require full harmonisation of the ETS design, it admits that certain design aspects have the potential to pose barriers compared to others, whereby any adjustments to an ETS could reduce technical and political acceptance to linking (Comendant and Taschini, 2014; Hawkins and Jegou, 2014; Pizer and Yates, 2015). One main recommendation is to harmonise the ETS design upfront to ensure market compatibility and to avoid disruption in the linkage. Some of the main factors to consider include the stringency of targets, enforcement, cost containment measures, eligibility of offsets and allocation methods (Jaffe, Ranson and Stavins, 2009; Metcalf and Weisbach, 2012; Zetterberg, 2012; Görlach, Mehling and Roberts, 2015). In the following section, these aspects are considered in detail, where different ETS design options relating to linkage are discussed.

2.3. International cooperation: framing concepts and design options for ETS linkages

In the future climate policy architecture, linkage is deemed to play a prominent role for the multiple opportunities and benefits of international cooperation via market instruments. Linking creates a common price for pollution, which equates marginal abatement costs between regulated jurisdictions, thereby increasing the access to abatement options and making it possible to attain the proposed mitigation target at a minimum cost to society. This overall reduction across participating jurisdictions occurs through the interaction of regional carbon regulations (Burtraw *et al.*, 2013; Bodansky *et al.*, 2014). This ultimately may lead to increased mitigation ambition and sustainable development.

Linking is considered to be a multifaceted political decision, agreed by participants in order to achieve environmental, economic and political goals. It is defined as a formal agreement by separated GHG mitigation programmes in different political jurisdictions (regional, national, or sub-national government) with the aim of maximising emission reduction efforts cost-effectively, and can take place among a heterogeneous set of policy instruments, involving non-market regulatory systems and/or market-based mechanisms, such as carbon taxes and ETS systems (Mehling, Metcalf and Stavins, 2017)[27]. The literature to date mostly focuses on linkage heterogeneity, in terms of policy instruments, political jurisdiction and targets (Lazarowics, 2009; Metcalf and Weisbach, 2012; Bodansky *et al.*, 2014; Mehling, Metcalf and Stavins, 2017).

Yet, such heterogeneous linkages are beyond the scope of this thesis, which alternatively focuses on linking carbon trading strategies. The literature to date has mostly focused on this way of linking (Görlach, Mehling and Roberts, 2015). In this type of coordination, the governing rules are adopted through a formal international treaty to bind the partners to domestic implementation, or

[27] Assuming that not all countries will employ an ETS to reduce emissions, Mehling, Metcalf and Stavins (2017) discuss the interaction of alternative policy instruments in a context of heterogeneity, which also applies to the level of political jurisdictions and types of targets. In addition, the authors bring practical lessons from evidences on linking heterogeneous climate policies. Additional qualitative information on this type of linkage can be found in Metcalf and Weisbach (2012).

through reciprocal legislation at national level which is followed by a non-binding expression of intent, for instance, an informal memorandum of understanding (Mehling, 2007). Among them, Mehling and Haites (2009) suggest that the binding agreement provides greatest certainty since it determines the exact legal conditions, including withdrawals and termination.

The linkage literature is usually separated into two main categories, theoretical/qualitative research and applied/quantitative research. Generally, the first category explores the advantages and disadvantages of linking, differentiating the system design features and other institutional aspects for enhancing the governance of the integrated system. Based on this theoretical foundation, the second brings about the practical aspects of linking, which include empirical exercises to analyse the policy implications. From the literature, important insights associated with the regulatory, economic and environmental aspects arise to help accommodate this strategy in the global policy framework.

Conceptually, a link can take either a direct or indirect form[28]. In direct linkages, emission allowances can be traded in one or more directions (unilateral, bilateral or multilateral connection) and are recognised for the purposes of compliance with the local cap. Under a unilateral link, one jurisdiction can buy emission allowances issued by the other but not vice versa. If the small systems are unilaterally linked to large ones, the effect on price of the large schemes is minor (Flachsland, Marschinski and Edenhofer, 2009b). In this type of linkage, considerable uncertainty in capped entities may arise, since

[28] Alternatively, another type of cooperation via linking that can be proposed, is known as clustered carbon market clubs (Espagne, 2015; Nordhaus, 2015; Victor, 2015) or instead, what is referred to as “linking by degrees” (Burtraw *et al.*, 2013). A likely feature of the Club would be the mutual access to the carbon markets (Petsonk, Keohane and Samans, 2015). The idea of forming Climate Clubs tends to lure policymakers, due to the potential benefits provided by wider diversity strategies available such as scientific cooperation, trade partnership and alliances for enhancing innovation and climate initiatives. Further, it can involve different economic actors into a common cause, for example, countries, cities, companies and NGO’s, into a broaden participation. The positive effect of joining a Climate Club in the context of international climate negotiations in the first place is to support fulfilling the NDCs but it may originate a rebound effect of boosting ambition levels in the Club (La Rovere, 2016). In the practice of linking by degree, an incremental alignment of key elements takes place prior to the potential introduction of formal linking, aiming at anticipating and addressing undesirable implications.

the amendments or the termination of the agreement is possible in light of developments in the other scheme.

On the other hand, a bilateral link enables full alignment so that allowances are mutually interchangeable and valid for compliance in both[29], with a full equalisation of prices. When the linkage integrates more than two systems, it becomes a multilateral link (Mehling and Haites, 2009). Indirect linkages[30] involve accepting allowances from a common third party, for example offset credits from the Clean Development Mechanism (CDM). Here, the discussion is limited to direct bilateral linkages, even though the use of offsets is part of the discussion on ETS design harmonisation.

There are different motivations for linking ETS systems. Table 2 summarises potential additional advantages as well as disadvantages of linkage. From the economic perspective, it promises further economic efficiency. As discussed in section 2.2.2, by expanding the scope and coverage a greater diversity of sources and mitigation options become available. In turn, the linkage leads to a reallocation of abatement efforts between regulated entities, thereby reducing emissions where they are least expensive (Anger, 2008; Tuerk *et al.*, 2009; Dellink *et al.*, 2014; Kachi *et al.*, 2015). This means that the extent to which more systems are linked, the greater the potential efficiency gains (Flachsland, Marschinski and Edenhofer, 2009b). According to Comendant and Taschini (2014), efficiency gains depend on overall market liquidity, which ultimately is affected by the economic shocks.

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[29] In direct linkages, allowances can be considered in a one-for-one basis (an equivalent unit in both systems) or instead, a trading ratio (an exchange rate). Despite facilitating combining two different levels of ambition, setting exchange rates for emission transfers may raise concerns on the economic benefits of linking, since it may prevent price equalisation across participants (Schneider *et al.*, 2017).

[30] In the indirect market, variations in supply and demand affect both systems regardless of being directly connected, and there may not necessarily be a complete convergence of the allowance price Flachsland, Marschinski and Edenhofer (2009b).

Table 2 – Potential advantages and disadvantages of linking ETS

Advantages	Disadvantages
<u>Economic:</u> 1 – Improve market liquidity 2 – Reduce price volatility 3 – Gains from trade 4 – Competitiveness: by reducing leakage <u>Political:</u> 1 – Signalling of multilateral effort for emissions mitigation 2 – Enhance domestic political acceptance <u>Administrative:</u> 1 – Reduce administrative costs and compliance costs <u>Environmental:</u> 1 – Achieve an aggregate reductions of emissions	<u>Economic:</u> 1 – Adverse distributional impacts 2 – Exposure to other market shocks 3 –Potential of expanding caps to increase permit sales <u>Political:</u> 1 – Risk of free rider behaviour 2 – Risk to endorse reduction targets less consistent with the socially efficient in a global perspective 3 – Deviation of policy objectives due to incompatible designs 4 – Reduced regulatory sovereignty <u>Administrative:</u> 1 – Transaction costs, mostly impacted by the scope of the system and choice of point of obligation <u>Environmental:</u> 1 – Reduced environmental performance: introduction of “hot air” and double counting

Source: prepared by the author from literature review.

However, Kachi *et al.* (2015) and Sterk, Mehling and Tuerk (2009) highlight that efficiency gains from linking two ETS systems can be limited if abatement costs are similar since the degree of economic efficiency depends on the heterogeneity of abatement alternatives. Therefore, the more diverse the market, the more options for emissions abatement with different associated costs. In theory, for a greater economic efficiency gain from allowance trading, the difference between the pre-link allowance price and the linked one plays an important role. Literature points out that the larger the difference in equilibrium allowance prices, the greater the potential gain in economic efficiency (Tuerk *et al.*, 2009; Dellink *et al.*, 2014; PMR and ICAP, 2016). Due to price harmonisation, competitive distortions that might arise from ex-ante price differences are also eliminated (Sterk, Mehling and Tuerk, 2009).

In fact, as allowance prices fully converge when linking up the markets, carbon price differentials are consequently eliminated. In general, ETS systems with a high permit price seek to link with systems with a lower price, and vice versa (Comendant and Taschini, 2014). An ETS where the emission price is higher before the linkage, benefits from the agreement once it tends to buy emission allowances from the other jurisdiction, reducing its price of compliance. In some cases, a lower carbon price may be undesirable, for instance, if the jurisdiction use the ETS to establish domestic price incentives for long-term investments in low-carbon infrastructure or in technological innovation (Calel and Dechezleprêtre, 2012).

Such welfare distribution across regulated entities matters (Newell, Pizer and Raimi, 2013). The policy structure, especially the harmonisation of permit prices, creates winners and losers in the link and political pressure, particularly from the latter, to drawback from negotiating a link is possible. Mostly, the challenge posed for institutional feasibility is related to the reduced control over prices, emissions and other aspects of policy design and impact (Ellerman, Buchner and Carraro, 2007; Ranson and Stavins, 2013).

Notwithstanding this, the price differential reflects the level of ambition or view about a desirable price signal, which makes the linkage challenging (Fankhauser and Hepburn, 2010) and is a determinant on the economic benefits to be obtained, as well as on the distributional impacts. This explains why the pre-selection of a linking partner is fundamental so to identify whether the differences are prohibitive to linking and to verify the compatibility of emissions cap and allowance price trajectories (Comendant and Taschini, 2014). Finding the ideal climate policy design before linking up is also relevant (Wagner, 2014).

In addition to reduced abatement cost and effects on allowance price, linking also promotes increased market liquidity, due to market enlargement (Grüll and Taschini, 2011; Metcalf and Weisbach, 2012; Ranson and Stavins, 2012), which provides the largest economic benefits from linking (Jotzo and Betz, 2009). More liquidity means that the linkage system has a reduced potential for affecting market prices through allowance trading. In theory, with more players and allowances in the ETS link, there is absorption of price variations and shocks from one system, being cushioned by the larger overall market (Zetterberg, 2012). The dual benefit lies in providing liquidity along with price stability, due to a reduction in price volatility caused by unexpected shocks. Yet, a recent study by Doda, Quemin and Taschini (2018) found that in a multilateral linkage, the price volatility is reduced on average, but not necessarily for all individual linkage group members.

Linking ultimately contributes to the avoidance of market power and price manipulation from larger entities. Nevertheless, price volatility can be transmitted from one system to another within the linkage (Jaffe and Stavins, 2007; Flachsland, Marschinski and Edenhofer, 2009b; Mehling, 2016). Evidence in the literature suggests that this is particularly relevant for small systems with relatively few participants, as linking with larger systems seems to have contributed to lower fluctuations in price, for example the one-way link of Norway and the EU ETS (Ranson and Stavins, 2012).

From the environmental perspective, there are arguments supporting linking ETS systems. First and foremost, with a broader participation and coverage, a larger share of global emissions is subject to reduction targets, which can be achieved in a cost-effective way. Depending on the stringency of caps, linking may further increase environmental effectiveness (Görlach, Mehling and Roberts, 2015) by signalling an international commitment to a long-term climate policy, where stringent targets may turn to be politically acceptable. This is a strong political dimension associated with the linkage approach. In fact, the decision of linking up is itself a demonstration of commitment to global action on climate change. For Flachsland, Marschinski and Edenhofer (2009a), linking may enhance further cooperation between nations and provide an example to follow, especially if a transatlantic linkage is agreed.

On the other hand, there are some concerns that linking could introduce a perverse incentive to reduce emissions abatement effort in installations or jurisdictions due to a more relaxed cap as a greater total number of allowances are available (Sterk, Mehling and Tuerk, 2009). Hence, compared to the case without linking, the resulting higher aggregated emissions from a less stringent cap create additional revenue through international trading (Helm, 2003; Calzadilla, Rehdanz and Tol, 2011; Ranson and Stavins, 2013). In this sense, Bodansky *et al.* (2014) and Stavins (2015) point out that instead of achieving environmental effectiveness, linkage can undermine the environmental integrity of the combined system. For instance, in the case of double counting or different levels of ambitions (Carbone, Helm and Rutherford, 2009).

An additional advantage is the potential for reducing the risk of emissions leakage, especially if the linking jurisdictions are also trade partners (as exposure to international competitiveness can be reduced). The price signal diminishes incentives to shift production elsewhere in order to avoid the emission regulation. In this sense, competitiveness concerns that might exist between the covered sectors before the agreement may be alleviated with the linkage. Some empirical research indicates the modest effects of climate policies on competitiveness, if it is not very strict (Dechezleprêtre and Sato, 2017), or if the scope is very limited (Flachsland, Marschinski and Edenhofer, 2009b). A study of Lanzi *et al.* (2013) reveals that in the presence of multiple ETS systems, the more emissions sources and countries covered, the lower negative sectoral competitiveness and leakage effects, especially when climate policies are harmonised across countries.

Literature in this regard suggests adhering to a global ETS to ensure that relative prices for all carbon-intensive products are consistent among all countries through the single carbon price worldwide (Baranzini *et al.*, 2017). As a result, a comprehensive worldwide system could guarantee the absence of carbon leakage (Babiker, 2005; Stavins *et al.*, 2014) and promote cost-effectiveness of mitigation at the global level. According to Bosetti *et al.* (2013), only a global cooperation to reduce emissions could maintain GHG concentration below 550pm CO2eq.

This is in accordance with Ranson and Stavins (2013), who argue that, in principle, a top-down negotiated global agreement could produce the best solution to the problem of climate change. Although there are differentials in marginal abatement costs and baseline GHG emissions, and a vulnerability to climate change across all nations; it could reduce global emissions to an appropriate level where each nation would be at least as well off than in the absence of global mitigation. Similarly, Flachsland, Marschinski and Edenhofer (2009a) found that top-down approaches tend to yield a higher degree of environmental effectiveness than bottom-up approaches, although a significant share of global emissions could still be captured by combining ETS systems. In addition, a single global market and carbon price appears to be more resilient to regional disruptions, spreading any imbalance over a larger volume of supply and demand (Pizer and Yates, 2015).

Nevertheless, differences in energy prices and income levels may limit the realisation of benefits, implying that local conditions need to be taken into consideration as well (IEA, 2005). In this sense, negotiating and designing a global ETS is very challenging in political and institutional terms (Baranzini *et al.*, 2017). Unsurprisingly, the real progress towards successful global climate coordination has been relatively slow. Irrespective of how recommendable a global participation might be, its likelihood or feasibility is very limited in the short term. In the absence of an international agreement, linked ETS systems at regional, national and subnational level could provide a bottom-up alternative, being a precursor to, and a stimulus for, a top-down future ETS approach, especially within the Paris Agreement architecture, as discussed in Chapter 3. To the extent which the number of linked ETS systems increases, the greater the appeal for a global international ETS (Haug, Frerk and Santikarn, 2015; Merkel and François, 2015).

This is particularly supported by the European Commission (2013), which envisages building a network of linked ETS. The European Union, through the EU ETS, offers a good example of leadership in the climate change policy arena, which intends to use the linkage as “a carrot” to encourage the development of ETS systems in other jurisdictions (Zetterberg *et al.*, 2012). Based on the lessons from the EU ETS, many systems have been planned worldwide. The strategy of linking to existing or emerging systems also demonstrates Europe's ability to support global cooperation on climate change.

The administrative benefits for potential partners are significant, since there is a potential for sharing of knowledge regarding the design and operation of the system (Stavins, 2015). Whilst linking harmonises the administration and design features of the ETS; it also allows improvements in the compliance process and can generate reduced administrative costs for regulated entities in both jurisdictions (Doda and Taschini, 2017). For example, the Brazilian system proposed in this research would benefit from linkage with the EU ETS through the sharing of practices in programme administration, especially because the Brazilian ETS has not been developed yet. In this linkage, it would be very likely that an alignment of features would simplify compliance and offer a reduced administrative cost for both jurisdictions.

In light of that, developing ETS systems and respective linkages may be key mechanisms to help address climate change. Yet, the proliferation of linkages would only occur if the anticipated benefits outweigh expected costs. To evaluate those, some diverse effects are considered for each particular linkage. The key question about a bottom-up climate policy architecture based on bilateral linkages is whether it could succeed in reducing a sufficient quantity of GHG at a reasonable cost (Heitzig, 2013). Moreover, the climate landscape is very challenging in the sense that is codified in legislation and regulations, not being permanent, and requiring a consistent mechanism to address the volatility caused by differences in regional economic performance (Ranson and Stavins, 2013). However, the ultimate success of linking will depend on the participating jurisdictions and the design of the arrangements in each link, since differentials between partners may affect the policy outcomes.

Coping with the degree of heterogeneity that characterises climate policies is very challenging when engaging in international linkage (Mehling, Metcalf and Stavins, 2017). Hence, the aggregated cost savings from linking could be an important argument against the resistance of business organisations and the general public to link ETS systems. Further, it could be used as an opportunity to incentivise jurisdictions to commit to targets that are more ambitious, and to agree upon a linkage that can also generate funds to re-invest in more reductions. This climate finance provided by the ETS may be economically appealing for the price signal it provides to attract investments on sustainable infrastructure (Studart and Gallagher, 2016) or to promote clean technology investments and economic efficiency (Farid *et al.*, 2016; IETA, 2016), particularly for developing countries. Notwithstanding the greater degree of acceptance from developing countries to participate in carbon markets set by developed countries, Gavard, Winchester and Paltsev (2016) show that it depends on the allocation of permit revenues so to compensate for potential negative effects.

Generally, existing ETS governance frameworks should involve a well-coordinated linked system so as to enhance mitigation and to diminish linkage problems or inefficiencies from the heterogeneity of policies associated with local preferences. This is based on the notion that differences in the ETS design largely affect the compromises that linkage would involve (Hawkins and Jegou, 2014). This approach is based on a harmonised ETS design and counterfactual price, and ensures not only that the greatest degree of compatibility possible is achieved, but also that disruption of the linked system is minimised (Tiche, Weishaar and Couwenberg, 2016; Quemin and De Perthuis, 2017).

A certain degree of harmonisation of the main features of both ETS systems is therefore fundamental to achieve the goals across jurisdictions, even though it involves a significant effort from the individual systems to negotiate how to align the existing features. The greatest obstacle to linking tends to be the need to harmonise programs in advance of linking, as frequently differences in market design reflect different preferences that may be hard to reconcile (Pizer and Yates, 2015). At the same time, the alignment is important to facilitate the linkage and avoid adverse effects. That was the case during the recent linkage agreement between the EU ETS and the Swiss ETS. There had been a long negotiation on the scope and coverage alignment, particularly with respect to the inclusion of the aviation sector which was not previously regulated by the Swiss ETS. Considerations on these aspects, and others, with respect to the ETS design options discussed in the literature are presented in the following section.

2.3.1 Prospects for linking: ETS design issues

Several factors determine the decision to link schemes. Ranson and Stavins (2013) found evidence that geographic proximity is the most significant predictor of entering in a linkage. The authors argue that linkages resemble trade agreements and similarly, “jurisdictions located near to each other may have similar environmental goals and economic conditions and may have a history of mutually beneficial engagement on other issues” (p.8). Another important consideration for linking is legal compatibility, as there is some degree of formality involved, with implications for the legal nature and the procedural requirements of linking adoption. Hence, the legal framework establishes rules, principles and procedures constituting the legal order (Mehling, 2016). There are also potential distributional impacts and respective ETS design elements influencing the decision to link.

In order to understand potential opportunities and risks of a certain link, Beuermann *et al.* (2017) define a systematic upfront assessment of the linkage based on three stages, assuming that potential linking partners have already been selected: i) pre-assessment, which means identifying and prioritising the most important objectives to be achieved along with the risks to be minimised; ii) an analysis of whether both schemes are similar or can be easily harmonised with regard to critical ETS features and the most likely ETS outcome and effects based on selected criteria; and iii) to conclude on the benefits and costs of linking.

Generally, limited quantitative and qualitative data is available in an ex-ante assessment so the use of economic modelling is necessary. From that, policymakers may evaluate where alignment is needed and whether the merits of linking exceed the demerits based on the national priorities, which are also reflected in the design variations amongst them. This linking process involves substantial costs and efforts that could discourage it (Doda and Taschini, 2017). At the same time, a positive decision to link is likely to occur if in the “matching linking partner’ selection process”, as defined by Comendant and Taschini (2014), essential features are identified to be more or less aligned. In this process, some design questions take place, such as whether the link will be bilateral, or there will be offset credits, banking and borrowing in the system.

Yet, linking does not necessarily require all design features to be identical, which is also the case for potential linkages with the EU ETS, as revealed by Hawkins and Jegou (2014). However, rather than enhance environmental efficiency and effectiveness, some design aspects may impair the objectives of the scheme more than others (Sterk *et al.*, 2006). Burtraw *et al.* (2013) and Kachi *et al.* (2015) discuss the differences in the design of schemes and which elements must be reconciled in order to link. Similarly, Fankhauser and Hepburn (2010) and Mehling *et al.* (2011) distinguish the design features that may generate political and economic obstacles to linking.

Overall, the main common design elements that apparently pose a technical barrier to linking are scope, coverage and other differences in the design, such as the point of regulation and opt-in and opt-out provisions. On the other hand, there are dimensions that contribute the most to increased economic issues, design problems or political concerns about financial flows from trading, which are necessary to harmonise. These are differences in the relative stringency of targets, treatment of free allocation versus auctioning, price management measures, temporal flexibility rules and eligibility of offsets. These design features reflect the environmental ambitions and aggregate goals of any linked ETS system. Note that there are risks of not anticipated outcomes even when compatible systems are integrated, for example, in terms of exposure to developments in one system that are propagated throughout the other (Flachsland, Marschinski and Edenhofer, 2009b; Quemin and De Perthuis, 2017).

The scope and coverage differentials do not impede the linkage nor do they affect its environmental effectiveness, which could in fact increase opportunities for abatement across a range of regulated entities and as a result, improve economic efficiency (Comendant and Taschini, 2014). However, these could impact the national costs of ETS implementation and change the environmental performance of the system in the case of divergences in the MRV system across them (Baron and Bygrave, 2002). By studying the EU ETS-

Norway ETS case, Hawkins and Jegou (2014) show that albeit scope and coverage can slightly differ, some jurisdictions may only agree on linking if some harmonisation is involved. Still, a completely equivalent sector in two independently-designed schemes is rather unlikely because countries have differing emissions profiles and have to choose accordingly which sources to include (Metcalf and Weisbach, 2012).

Despite not necessarily implicating a negative environmental effect, differentials of scope and coverage may generate distributional impacts if the linkage occurs among heterogeneous systems (Metcalf and Weisbach, 2012). For instance, if the linkage is negotiated between a broader base ETS and a narrower one, the former may not be willing to link since the latter tends to present higher abatement costs and hence gains more from the linkage. In other words, linking systems that differ in the sectors or gases included may create winners and losers. This presumably might lead to a setback of the linkage process. For developing countries aiming to design an ETS and link it to existing developed-world programmes, this aspect has to be carefully negotiated.

Conversely to scope and coverage, it is apparent from the literature that bilateral linkages with differing stringencies of emissions cap is likely to be very challenging, if it is possible at all (Goers and Pflüglmayer, 2012; Burtraw *et al.*, 2013; Comendant and Taschini, 2014). In theory, different levels of ambition need not necessarily be an obstacle and an "allowance exchange ratio" could simply be implemented to address this issue (Burtraw *et al.*, 2013). In practice, however, experiences demonstrate that these differences pose a real barrier to linking. One example is the California opposition to develop a link with the EU ETS due to the collapse of allowances prices in the latter (ClimateWire, 2013).

In terms of stringency, the literature observes a relationship between trade of emissions allowances and cap stringency. In general, along with environmental goals, an ETS is designed to obtain economic benefits and a country may choose its cap strategically in order to maximise potential gains from future trading (Helm, 2003). The cap's design translates the level of commitment of each linked ETS system ergo, the environmental outcome desired. Cap and stringency requirements will vary according to an economy's size, nature and level of development which are difficult to coordinate when linking schemes from different jurisdictions. Green, Sterner and Wagner (2014) and Burtraw *et al.* (2013) consider this to be the most prominent barrier to linkage which is relevant to the functioning of the markets and the political economy of the ETS, particularly if the methodology on which the cap is based diverges.

There are some reasons for characterising the cap as a crucial aspect, especially in political terms, and whose agreement consists of a pre-requisite for successful linking. Firstly, diverging levels of ambition may signalise different views on which countries should pay for mitigation, with potential for rendering the link to be unacceptable. Secondly, it can lead to significant differences in allowance prices as well as the efficiency gains. Accordingly, there would be distributional effects and revenue transfers from linkage, where the less stringent system would face higher allowance prices and the more ambitious one a substantial financial outflow. Hence, when deciding to link, both systems need to be aware and comfortable with the resulting trading pattern (Comendant and Taschini, 2014). With the lack of stringency harmonisation, linking may raise equity concerns and even prevent the linkage from materialising (Green, Sterner and Wagner, 2014).

Although technically differences in the stringency of cap are possible, from the political perspective, this is not always clear. One example could be a cap set above the BAU emission level, since it would undermine the environmental effectiveness of linking (Sterk *et al.*, 2006). Also, where the level of ambition and marginal abatement cost widely differs, there is potential for a race to the bottom in the linked systems, which could become more pronounced if the linkage implies raising the cap, meaning less abatement for a given ETS (Zetterberg, 2012). In some cases, the ETS with more ambitious targets will attempt to persuade other systems with less stringent goals to adopt their tighter levels of abatement, thereby hesitating to link unless cap levels are harmonised with the accepted burden-sharing rule (Flachsland *et al.*, 2008b). Such incentives to alter domestic emissions caps when national ETS are linked are explored in Carbone, Helm and Rutherford (2009) and Holtsmark and Midttømme (2016).

Based on the abovementioned context, Ranson and Stavins (2013) argue that linking jurisdictions that share a similar ambition (i.e. comparable caps) is easiest, conditional on their relative levels of economic development. Haites (2014) and Edenhofer, Flachsland and Marschinski (2007) suggest comparable vision of medium to long-term emissions trends as relevant for linking.

Additionally, the cap structure plays an important role in the environmental effectiveness of the agreement. In theory, a bilateral ETS linkage between an absolute-based and an intensity-based cap system is not technically impossible, but it is complex and give rise to concerns in relation to competitiveness, cap integrity and liquidity shocks (Comendant and Taschini, 2014; Hawkins and Jegou, 2014; Kachi *et al.*, 2015).

Given these potential effects, absolute cap based jurisdictions may decline to link with those based on intensity caps. According to Fisher (2003), if the cap in one jurisdiction is set on absolute emissions whereas the other is based on intensity cap, international trading may result in an overall increase in emissions, compared to the situation with no linkage, thereby negating environmental effectiveness. A potential problem is that the level of production, and thus emissions' level of emitters covered by intensity caps are allowed to increase, which could be viewed as a transfer of welfare by the absolute-target system (Comendant and Taschini, 2014). As intensity-based allocations occur in two stages, that is, an initial allocation based on the projected output, and an ex-post adjustment after knowing the actual production levels, there may be liquidity shocks at the moment of adjustment (Sterk *et al.*, 2006).

Others design features are regarded as critical for linking up ETS systems. Literature mentions price management (or cost-containment) and the recognition of offsets as the most problematic for the agreement (Tuerk *et al.*, 2009; Zetterberg, 2012; Kachi *et al.*, 2015), in particular if a given ETS is concerned with the integrity of their emissions cap (Comendant and Taschini, 2014). These elements are designed to control the range of allowance prices in order to attract domestic support for the climate policy. Price support and price containment measures (price ceiling and price floor) have implications on both linked schemes, whereby some issues arise from the supply side of the allowances. Once those measures exist in one ETS it propagates to the other. If before the linkage, price management measures differ among the schemes, prices when the schemes are linked can also be affected. In a fully integrated ETS system, impacts on prices are more significant for smaller systems, frequently with no domestic price containment policies and are more exposed to the demand patterns imposed by larger systems (Comendant and Taschini, 2014).

In addition, there are also limitations for the link in the presence of temporal flexibility. Banking provisions of one system extend to the other and as a result, there is an increase in the volume of future emissions allowed in the overall linked system due to emission reduction of the ETS with banking. A non-harmonised banking or borrowing provision can generate price and investments distortions as well as low environmental effectiveness. For this reason, deciding the length of trading period is relevant for inter-temporal efficiency and investments strategies. Similar to the cap, calibrating the rule within the trading system is essential in order to attain the primary objective of mitigation.

Offset provisions may also be critical to linkage and demand a degree of harmonisation (Zetterberg, 2012; Burtraw *et al.*, 2013). Linkage allows the existing offset credits in one ETS to be available in the linking partner ETS (at least indirectly). In this sense, the amount of offsets allowable for compliance purposes, the type of offset which is eligible, the stringency of standards and the potential for double counting summarises the main concerns outlined by Kachi *et al.* (2015). By allowing offset credits to be interchangeable with allowances in one system, allowances that could have been sold to the other are liberated. For the environmental integrity of the linked ETS system, differences in the scope of offsets eligible for domestic compliance are very difficult to cope with, when mitigation goals may end up being (partially) fulfilled in a way that otherwise would be ineligible (Comendant and Taschini, 2014). In practice, it is observed in the EU ETS case that the use of offsets, which has so far been permitted, is not envisaged post-2020.

Conversely, there is a set of programme elements that do not represent a barrier to linkage, but alignment could help to further address environmental and competitiveness issues, besides contributing to more efficient operation, such as the point of regulation and opt-in and opt-out provisions, commitment period and enforcement provisions (Burtraw *et al.*, 2013; Kachi *et al.*, 2015). A certain degree of consistency between the linked systems is also required in terms of harmonising the participation status, whether it is mandatory or voluntary and the robustness of MRV and enforcement systems.

Under the allocation method perspective, differences defined separately in the ETS prior to linkage are not technically an obstacle (Tuerk *et al.*, 2009; Metcalf and Weisbach, 2012), as long as the cap is fixed (PMR and ICAP, 2016). Ultimately, the environmental effectiveness of the linkage should not be affected since it is determined by the overall cap. The co-existence of different allocation modes can cause competitiveness issues due to price changes, with potential to be viewed as unfair by some participants, especially those not granted a lump sum subsidy. In other words, the effects of how allowances are distributed, that is free distribution in one compared to auctioning in another, can also provoke competitiveness distortions (Ahlberg *et al.*, 2013). For Baron and Bygrave (2002), the effects of linking a grandfathering system with an auctioning system with regard to competitiveness depends on the design of each system, including the distribution of auction revenues.

To comply with such concern, Sterk and Mersmann (2011) state that the system based on auctioning is likely to demand an ex-ante harmonisation. This recommendation is in line with the literature on linkage, which supports an upfront negotiation of the alignment of the ETS features, as previously mentioned. The same logic is valid to facilitate a linkage between jurisdictions at different levels of economic development (Comendant and Taschini, 2014). In general, experience suggests that smaller systems with willingness to link tend to cede proprietary ETS design elements to ensure the necessary level of compatibility and regulatory convergence compared to larger ones (Mehling and

Görlach, 2016). At the same time, jurisdictions may be reluctant to revise the ETS features, making subsequent adjustments difficult because of path dependencies and the need to respect political compromises from the initial implementation process (Pizer and Yates, 2015).

In order to avoid reducing regulatory sovereignty, it is important to develop a robust regulatory framework that appropriately accommodates the remaining differentials so as to prevent negatively affecting the functioning of the linked system over time. Another particular relevant governance arrangement to comply with the evolvement of the linked ETS is the inclusion of a contingency plan for delinking, as ETS linkages are not necessarily permanent. Implications of delinking are evaluated in Pizer and Yates (2015). In this context, the agreement needs to be structured considering adjustments of the cap, the treatment of allowances from another system and a clear exit strategy, which is a critical aspect in links where jurisdictions do not have a close engagement in other issues (PMR and ICAP, 2016).

The proposal analysed in this thesis is based on the literature regarding the advantages and potential challenges of linking ETS systems described in this chapter. Whilst, on the one hand, there is significant motivation to link domestic schemes, some concerns have to be addressed before moving forward with such a proposal. Brazil and other developing countries have just recently committed to climate change mitigation with mandatory reduction pledges under the Paris Agreement. For those countries, evaluating the appropriate design of the scheme’s main features and deciding strategically on the elements to be harmonised in the link is essential. This approach will help lead to political acceptability, whilst at the same time avoid negative distributional effects and ultimately help to achieve the expected environmental and economic benefits.

Several topics of the climate agenda have been discussed internationally with the objective of obtaining a mutual agreement. The focus of the next chapter is to provide a brief introduction on the role of international negotiations in relation to ETS strategies and the relevance for adhering to market-based cooperation post-2020, with emphasis on Brazil, Europe and potential alternative partners, notably Latin America and China.

CHAPTER 3

INTERNATIONAL EMISSIONS TRADING AND LINKING STRATEGIES POST-2020

3.1 Introduction

Given the urgency to undertake ambitious efforts to combat climate change and adapt to its effects, more than 180 Parties of the United Nations Framework Convention on Climate Change (UNFCCC) met in Paris from 30 November to 12 December 2015 to negotiate a new global and legally binding deal in the 21st Conference of the Parties (COP-21). Previous international negotiations had achieved little progress, which brought about significant discussion on the importance of setting up a climate pact in which all countries were committed to.

There appears to be consensus on the potential impacts of carbon emissions in the global climate since the late 1980's, when the Intergovernmental Panel on Climate Change (IPCC) was created with the aim of providing a review and assessment of the most recent scientific, technical and socio-economic views on the topic. The first IPCC[31] Assessment Report in 1990 brought to light the breadth of climate impacts, as well as the need for a political platform at international level, leading to the establishment of a framework of climate protection governance in 1992 at the Rio de Janeiro Earth Summit (Rio-92). Building the UNFCCC sets a remarkable global attempt to control GHG emissions' growth, whose main objective was to "achieve, in accordance with the relevant provisions of the Convention, the stabilization of GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system" (UNFCCC, 1992, p. 4). If attaining such level within an adequate timeframe, ecosystems could adapt naturally to changes in climate, whereas food production and economic development could proceed in a sustainable manner.

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[31] IPCC operates under the United Nations Environment Programme and the World Meteorological Organization support. Up-to-date reports summarising climate change science, quantifying effects and evaluating policies are released every six years.

Following Rio-92, yearly negotiations were initiated to set regulatory instruments in order to strengthen the commitments and hence impose stricter emission reductions targets on all participants (or Parties)[32]. The first successful policy architecture for global climate change was the Kyoto Protocol, when discussions began in 1995, proceeding to adoption in 1997 and conclusion in 2001 at COP-7 in Marrakesh, when all the rules for implementation were defined, including the structure and institutions to support achieving the long-term objective.

Among the 192 UNFCCC Parties who ratified the Protocol (excluded the US and Australia), only developed countries listed in Annex I were subject to a legally binding commitment, with a target of 5% emissions reduction against 1990 levels in the first period of compliance (2008-2012), and 18% in the second period (2013-2020), which were considered relatively modest (Falkner, 2016). Under the Kyoto Protocol, developing countries were exempted from similar commitments based on the “common but differentiated responsibilities” principle, on the basis that developed countries must face the burden of their historical releases of GHG emissions into the atmosphere as a result of industrialisation. Adequate financial and technological resources are available in those countries to adjust emissions levels towards a low-carbon economy, besides supporting developing countries in the adaptation process and development trajectory.

One major effect of such differentiation is that mitigation by developing countries, even the larger ones such as China, India and Brazil, was postponed. The main argument to prevent accepting the adoption of a binding mitigation commitment was due to other urgent priorities such as development and poverty reduction. At the same time, these developing countries have been actively involved in climate mitigation through the baseline-credit-system originated from the Kyoto Protocol, specifically by hosting Clean Development

[32] COP is the annual meeting organised to regulate, discuss and implement actions, particularly in terms of international cooperation for climate change mitigation and adaptation.

Mechanism (CDM) projects[33], or in some cases, by committing to voluntary emissions reductions.

Article 17 of the Kyoto Protocol introduced three flexibility provisions to help reduce compliance costs of Annex I Parties. One of them is the ETS, which enables international trading of Assigned Amount Units (AAUs) at country level within the commitment period. This inclusion was inspired by the US success in addressing acid rain with the SO_2 ETS. Further, it allows for project-based credits generated through CDM projects undertaken with a developing country or Joint Implementation (JI) projects with another developed country, to be valid for compliance. In this case, certified emission reductions (CERs) and emissions reduction units (ERUs) issued for each project approved, are tradable in the existing carbon markets of developed countries, or could be used to offset an emitting activity or used as a voluntary measure (Carraro and Favero, 2009).

With this architecture, Falkner (2016) argues that the Kyoto Protocol has, however, failed in offering a viable approach to decarbonise the economy as little dynamic incentives were created. In addition, strengthening the commitments in the second period has proven to be difficult since those whose mitigation burden was largest become reluctant to set additional rigid targets, coupled with a sense of inequity due to a lack of binding commitments for developing countries with increasing levels of emissions. A study of Böhringer and Löschel (2003) indicated a lower aggregate reduction burden on industrialised countries of 15% relative to BAU when developing countries commit to cut 5% of emissions. However, the acceptance of mitigation targets leads to a worsened welfare situation for developing countries. This is supposedly one concern and reason to postpone climate action in these countries.

Even though the Kyoto Protocol's architecture alone could not guarantee the emergence of an efficient ETS at international level (Baron and Bygrave, 2002), it has at least succeeded in laying the ground for a new approach for

[33] Following an independent line in the negotiation, Brazil proposed the CDM (Viola, 2004).

global mitigation, after many years of negotiations (started in the 2009 Copenhagen Conference, COP-15). Culminating in the first truly global international treaty, the Paris Agreement, as it became known, brings together all countries into a common cause so that a low carbon, resilient and sustainable future may be feasible, as originally advocated by the UNFCCC. The bottom-up framework involves a global long-term action plan committed to keep global warming well below 2°C and makes efforts to limit the temperature increase to 1.5°C by 2030.

The new paradigm consolidated in COP-21 recognises the historical, current and future responsibilities of the Parties, including developed and developing countries, signalling to the collective need to switch from the consumption of fossil fuels to clean sources of energy. In terms of international climate geopolitics, this is an important achievement since major developing countries are, for the first time, formally committed to global mitigation. Further, there have been improvements in terms of putting in place appropriate financial flows, technology frameworks, and capacity building as well as a more robust transparent system to support action undertaken by developing countries, especially the most vulnerable.

The Paris Agreement was entered into force earlier than expected on 4th November 2016. It will become effective from 2020 when approximately 105 Parties[34], accounting for more than 55% of total GHG emissions, have deposited their instruments for the agreement ratification.

There would not appear to be general consensus on the potential of the agreement to promote sufficient positive outcomes. On one hand, some studies emphasise that the multilevel climate governance of the agreement represents a diplomatic and political success (Dimitrov, 2016; Sirkis, 2016; Soto, 2016). On the other hand, there is limited progress on certain elements not fully assigned in the agreement, such as the lack of enforcement mechanisms, or provisions for compensating loss and damages caused by extreme weather events (Vieira

[34] The US president has recently publicly announced the intention to withdraw the USA from the agreement, which may pose serious risk to the common objective of tackling climate change and its effects on the planet.

and Vernet, 2015; Barata and Kachi, 2016). Under this perspective, these aspects may lead to a misleading climate goal, predominantly focused on economic development (Boff, 2015).

In fact, the document establishes the climate goals and the framework for international climate action, but does not specify the details in depth. Most of the specifications were postponed until 2018, as decided during the first conference session of the Paris Agreement in Marrakesh (COP-22). For example, a decision on the rules governing market and non-market mechanisms that have been set up under the accord's Article 6, which provides the opportunity to expand the reach of carbon pricing, still have to be negotiated over the coming years. In COP-23, a number of topics on the scope and governance are to be defined, such as the use of sectoral approaches or the REDD+ activities (Reducing Emissions from Deforestation and Forest Degradation, and sustainable forests and enhancement of carbon sinks), as well as the need to place any restriction on the cooperative approach and the use of Internationally Transferred Mitigation Outcomes (ITMOs) (World Bank, Ecofys and Vivid Economics, 2017).

Although not explicitly referenced as a market-based approach, the provisions introduced by Article 6 allow the use of international transferred mitigation outcomes to comply with the Nationally Determined Contributions (NDCs). These NDCs have been prepared for each Party in order to publicly communicate formal obligations, including the future trajectory of emissions and domestic mitigation measures for achieving them. In order to facilitate overall emissions abatement and enable higher ambition, different provisions were incorporated into the plan for increasing flexibility as described in Article 6: a cooperative mechanism based on a top-down voluntary cooperation for allowing the trading of ITMOs; an alternative mechanism to contribute to mitigation and support sustainable development substituting the previous Kyoto flexibility system; and finally, non-market measures.

We focus the investigation on the cooperative approach as a mean to help participants meet annual emissions reductions. Most characteristics of the cooperative perspective are identified at articles 6.2 and 6.3 of the Paris agreement, which includes specifications for the voluntary nature of the cooperation, how it involves the use of ITMOs towards the NDC, whilst also ensuring environmental integrity and transparency along with robust accounting[35] (UNFCCC, 2015a). The voluntary basis of a cooperative agreement reflects the idea that the market provisions structure must only be imposed on those adhering to the integrated system (Barata and Kachi, 2016).

According to the cooperative perspective, international transfers may be conducted by linking ETS systems or other national climate policies. In addition, other types of cooperation related to the existing elements of Article 6 can be adopted, such as clustered carbon market clubs (Espagne, 2015; Nordhaus, 2015), or even a more centralised approach via the UNFCCC. In a two-way ETS linkage for climate cooperation, ITMOs are mutually interchangeable and accepted for compliance purposes, with a full equalisation of prices. If country A is unable to easily curb emissions at lower cost, the alternative is to acquire additional units from country B, who finds it relatively cheaper to undertake extra abatement or to invest in new technologies for mitigation, and benefits by selling unused units. The financial flows would go from A, the buyer of units, to B, which invested in emissions abatement. From this trading transfer, responsibilities may potentially be allocated and yield gains realized for countries that otherwise would not benefit from committing to climate mitigation (Aldy and Pizer, 2009).

As previously discussed, in theory both countries benefit from the linkage: from the cost savings of emission reductions to the potential maximisation of revenues from enhanced carbon trading opportunities. IPCC (2001, p. 607) highlights that the benefits of cooperation rely on "the potential to address several challenges: multiple actors that are diverse in their perceptions of the costs and benefits of collective action; emissions sources that are

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[35] In the context of climate mitigation targets, the term "accounting" refers to a framework that makes mitigation commitment and progress comparable, in order to evaluate achievability of targets (Prag, Hood and Barata, 2013). Thus, robust accounting properly quantifies anthropogenic emission changes, according to the sources, or removals by sinks as a result of mitigation actions by countries or other entities.

unevenly distributed; heterogeneous climate impacts that are uncertain and distant in space and time; and mitigation costs that vary".

On the other hand, Babiker, Reilly and Viguier (2004) underline that not all countries benefit equally from the introduction of an international ETS system, given the pre-existing institutional environment and the terms of trade effects. Some other research has alleged that ETS systems have a limited capacity to drive positive climate outcomes. Sirkis (2016) argues that an ETS is limited in scope, which is set to rationalise the achievement of already established goals, not necessarily triggering a deep decarbonisation of the economy. In this context, and due to some financial incentives, it may encourage those participants not willing to reduce emissions, to make less effort to do so, or to not reduce emissions at all (Lohmann, 2006). Fundamentally, the critical problem of adhering to market-based instruments and expanding their reach stems from the commodification[36] of emissions since it can exacerbate social and environmental inequalities[37] through unequal distribution of income, and unequal exposure to the negative ecological effects of economic activity[38] (Böhm, Misoczky and Moog, 2012). Irrespective of how applicable the market mechanisms are, there is no guarantee of avoiding further negative effects on the environment, or being consistent with sustainability.

Although this controversy over the role of markets in international climate cooperation exists, it is important to recognise that the number of existing ETS systems worldwide is on an upward trend and may become more common post-2020, as 91 Parties declared an interest to access international markets for mitigation purposes, including both developed and developing regions (IETA, 2016). As a consequence, the more interest in implementing carbon trading mechanisms the more linkages are expected to emerge among participants.

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[36] In summary, the commodification concept is referred to as the institutional, symbolic and material changes through which a good or service that was not previously meant for sale gets into the market exchange arena (Bakker, 2010; Kallis, Gómez-Baggethun and Zografos, 2013).

[37] According to Lohmann (2009), inequalities are magnified by carbon pricing mechanisms for several reasons. Firstly, by creating transferable rights to pollute and awarding those to large emitters, it allows the ecosystem capacity to be exceeded and generates disproportional effects on small islands, coastal and local communities, as well as the poorer. Further, the refusal to phase out fossil fuel energy tends to cause increasing conflict around the world.

[38] In this thesis, we will not focus on this aspect specifically.

Under a bottom-up policy architecture, this mechanism has the potential to reinforce the realisation of NDCs, thereby being crucial for the cost-effectiveness of the Paris Agreement (Mehling and Görlach, 2016; Doda, Quemin and Taschini, 2018). Most importantly, it is likely to enhance the understanding and acceptance of the concept of an ETS to form the basis for a global approach.

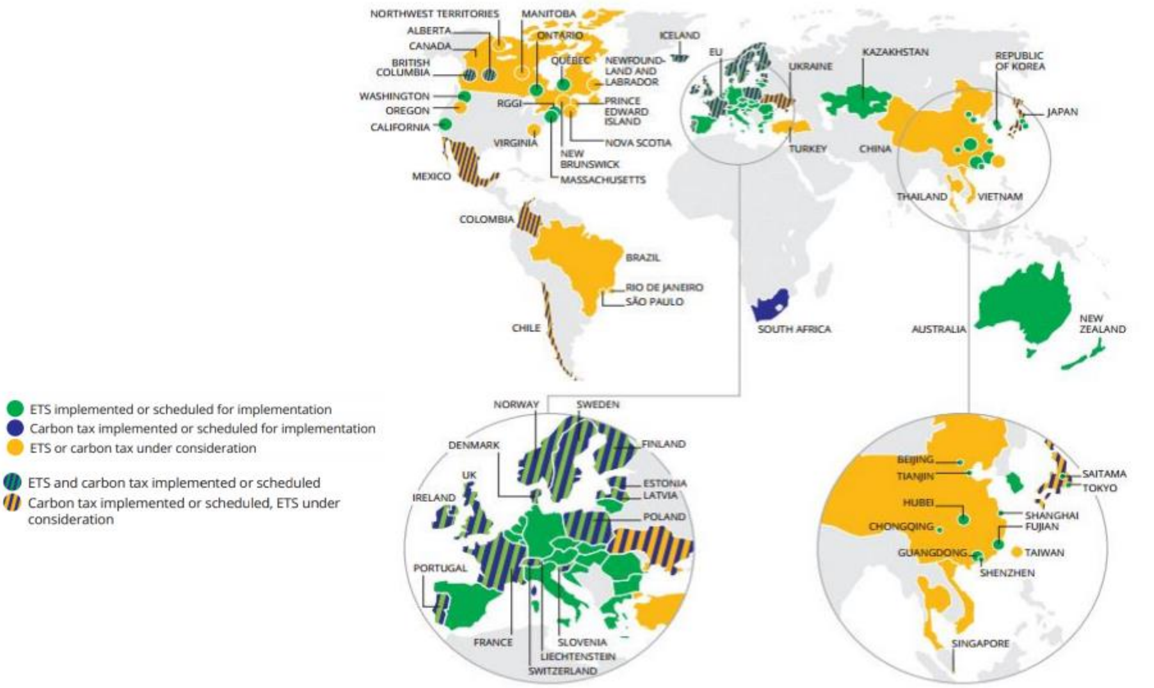
Therefore, in the future global climate policy framework, linkages at different levels could be a fundamental element (European Commission, 2015b; Doda and Taschini, 2017), where an increasing appeal may develop for a global international ETS (Haug, Frerk and Santikarn, 2015; Merkel and François, 2015). In the literature, a comprehensive worldwide system is considered to be effective, at least in terms of climate protection and access to less costly abatement options (Flachsland, Marschinski and Edenhofer, 2009a), although mitigation costs may differ across regions (Luderer *et al.*, 2012).

Nevertheless, irrespective of how recommendable a global participation might be, its likelihood or feasibility is very limited in the short term. For this reason, linked ETS systems could be viewed as a precursor to, and a stimulus for, a top-down global ETS approach, especially within the Paris Agreement architecture. Despite existing concerns over the potential economic impacts, political preference has emerged around the world for implementing ETS systems for achieving GHG emission reductions (Schmalensee and Stavins, 2017b).

3.2 Linking ETS systems around the world: an empirical perspective

The most up-to-date report of the World Bank (World Bank, Ecofys and Vivid Economics, 2017) outlines the existence of 42 national and 25 subnational jurisdictions with carbon price initiatives implemented, as depicted in Figure 2. Those initiatives account for about 50% of the global economy and more than a quarter of global GHG emissions. Also, the report shows that over the last five years, half of the new initiatives initiated or scheduled for implementation were in upper-middle-income economies, conversely to the overall trend prior to 2013. In the last two years, Latin America has drawn attention, where six newly implemented initiatives took place, totalling 12 in the 2016-2017 periods, and with potential to double in the future. Overall, these numbers demonstrate the strong momentum for carbon pricing around the world (Rydge, 2015).

Figure 2 – Carbon pricing initiatives worldwide



Source: World Bank, Ecofys and Vivid Economics (2017).

There are ETS systems in force at several levels of governance, including in subnational jurisdictions in Canada, Japan and the USA. Amongst them, there are substantial differences in terms of size, design characteristics, geographical scope, or other aspects such as the point of regulation, the nature of the ETS (if mandatory or voluntary) or the period of compliance.

Table 3 – ETS systems in force at national level

ETS characteristics	European ETS	Swiss ETS	New Zealand ETS	Korean ETS	Kazakhstan ETS
Implementation	2005	2008	2008	2015	2013
Sectoral Coverage	Power, Industry and Aviation	Industry	Power, Industry, Buildings, Transport, Aviation, Waste and Forestry	Power, Industry, Buildings, Aviation, Waste	Power and Industry
Emissions Coverage	1.839 $MtCO_2e$ (CO2, N2O, PFCs)	5.1 $MtCO_2e$ (CO2, NO2, CH4, HFCs, NF3, SF6 and PFCs)	41.7 $MtCO_2e$ (CO2, CH4, N2O, SF6, HFCs and PFCs)	538.5 $MtCO_2e$ (CO2, CH4, N2O, PFCs, HFCs, SF6)	161.9 $MtCO_2e$ (CO_2)
Point of Regulation	Downstream	Downstream	Upstream	Downstream	Downstream
Carbon price	EUR 9.68 (approx. USD 11.92)	CHF 7.50 (approx. USD 7.79)	NZD 21.10 (approx. USD 15.47)	KRW 22,000 (approx. USD 20.66)	No information (it has just restarted operation)
Allocation	Free allocation and auctioning	Free allocation and auctioning	Free allocation	Free allocation	Free allocation
Flexibility provisions	1.Unlimited Banking Borrowing not allowed 2.International offsets allowed by the end of Phase 3 3.Price containment measure (Market Stability Reserve)	1.Unlimited Banking 2.International offsets allowed under some conditions	1.Banking Domestic offsets	1.Banking Domestic 2.International offsets allowed	1.Banking within trading period 2. Domestic offsets

Source: prepared by the author based on ICAP (2018).

At country-level, Table 3 describes the five existing systems, namely the EU ETS, the Swiss ETS, the New Zealand ETS, the Korean ETS and the Kazakhstan ETS, which has just recently restarted operations after two years of temporary suspension[39]. The EU ETS is the largest and most consolidated system in the world with 31 countries participating (all 28 EU countries plus Iceland, Liechtenstein and Norway), which account for 45% of the EU's emissions of the power, energy-intensive industries and aviation sectors. Following the launch of the EU ETS, the Swiss ETS started in 2008 as a five-year voluntary system as an alternative to the carbon levy on fossil fuels. It has become mandatory in the second phase for large energy-intensive industries, covering around 10% of the Swiss total GHG emissions.

In parallel, the first ETS in Oceania was launched in 2008, the New Zealand ETS (NZ ETS). Interestingly, it covers the majority of economic sectors, including forestry and waste, even though nitrous oxide and methane emissions from agriculture are subject only to reporting obligations. Note that forestry plays an important role in New Zealand, particularly carbon sequestration, justifying its first-hand inclusion in the ETS so as to provide accurate price signals to reduce deforestation and land use change (MAF, 2011). The NZ ETS case provides the only example of forest landowners under an ETS regulation, but also an example of delink from international markets of the Kyoto Protocol (Carver, Dawson and Kerr, 2017). There have been two statutory reviews that resulted in adjustments of design and operation in the NZ ETS for future compliance periods, notably the introduction of auctioning, new price ceiling measures and limits to the volume of international units.

The first nationwide ETS implemented in Asia is in Kazakhstan, the KAZ ETS. Considered to be an upper-middle-income economy by the World Bank, it is the largest economy in Central Asia. Under a framework based on the EU ETS, the KAZ ETS was implemented in 2013 aimed at switching from fossil fuels to cleaner and more efficient technologies, particularly in manufacturing

[39] According to ICAP (2018), there have been soft obligations during this period of suspension. Yet, amendments were approved within the Environmental Code in 2016 in order to enhance the MRV system, to improve the regulation and operation of ETS, for example by including benchmarking as the method of allocation.

and electricity generation (ADB, 2016). Given the number of ETS systems being considered or scheduled for implementation post-2020, the KAZ ETS has expressed an interest in future linkages.

Currently, there are other two operational ETS systems in large developing countries, namely in China and the Republic of Korea. The nationwide Korean ETS (KETS) is active since 2015, being in the second phase of the programme. Emissions coverage comprises 68% of national GHG emissions from power and industry sectors. Research by Choi, Liu and Lee (2017) has modelled the economic impacts of the KETS and found significant abatement effects combined with small negative impacts on GDP and household consumption for the 2020-2030 period. Moreover, the ETS enhanced competitiveness by providing a higher trade surplus. In light of the discussions on potential collaboration with New Zealand (World Bank, Ecofys and Vivid Economics, 2017), future linkages may further expand the benefits of the ETS.

After being beset by delays, the Chinese ETS was launched in December 2017, following some years of experience with subnational pilot markets. The learning-by-doing process was intended to be the first step towards adopting a market-based approach for mitigation, providing lessons for the design and operation of the national system. According to Goulder *et al.* (2017), a nationwide programme in China is very challenging due to the country's sheer size, industrial and geographic heterogeneity, income disparities and diverse institutional characteristics, as well as the substantial presence of state-owned enterprises in the natural gas and electricity industries, where these exert market power and prices are administered.

In order to understand how to cope with these issues via an ETS, five municipalities and three provinces with different economic structures and development levels were selected for the pilot phase of carbon trading (Duan and Zhou, 2017): Beijing, Chongqing, Shanghai, Shenzhen, Tianjin, Guangdong, Hubei and Fujian. The implementation of these programs was initiated in 2013, with coverage of the power and industry sectors, while the inclusion of other industries differed. Whilst in Shenzhen commercial buildings and road transportation are regulated, Beijing ETS includes hotels, universities and medical facilities (Goulder *et al.*, 2017). Similarly, domestic project-based carbon offsets credits are allowed in each pilot, with distinct limits applied. In the case of the Fujian ETS, a special focus is given on developing forestry projects to reduce emissions and trade offsets. Yet, the literature suggests that the success of these pilots ETS was mixed (Zhang *et al.*, 2014; Munnings *et al.*, 2016).

The legal framework for China's national ETS has been shaped by lessons from the pilots, and has been designed to address possible impacts on the economy with unified rules on the ETS design across the system. Covering only carbon emissions from approximately 1700 companies, the Chinese ETS is expected to become the largest in the world as it projects a gradual increase in the scope (the plan is to cover eight sectors: petrochemical, chemical, building materials, steel, nonferrous metals, paper and aviation). Details on the ETS design are laid out in the Work Plan for the construction of the National ETS. In short, it is designed to regulate the power sector (including combined heat and power, as well as captive power plants of other sectors) in order to meet the intensity targets committed in the Paris Agreement by 2030. Initially, the system accounts for only 30% of national emissions (ICAP, 2018). However, the plan is to eventually curb 50% of China's GHG emissions through the ETS in the future. Since China is currently responsible for about 30% of global emissions, its decarbonisation will largely help in addressing global climate change.

An initial three-phase roadmap has been adopted for the Chinese ETS, with an annual period of compliance. At first, the plan is to develop market infrastructure, followed by a simulation trading phase and by 2020 to deepen and expand trading. Further phases and allocation rules are yet to be defined. However, free allocation based on sub-sector benchmarks is expected during the trial allocation, as well as banking of allowances. Developments of the ETS are planned over the next few years to introduce offsetting mechanisms, basically the Chinese Certified Emissions Reductions (CCER), which is in line with the pilot programmes continuously active. Trading is expected to happen after 2020 and some pricing control rule such as auction reserves might be introduced (Tang, 2017).

Previous studies of the literature have examined China's commitment with mitigation and the importance for achieving an acceptable global temperature goal (Paltsev *et al.*, 2012), indicating that if China participates in a global regime, concentration of CO_2 could change between 200 and 280 ppm, the equivalent of 1.3 degree Celsius above industrial levels. The investigation of Hübler, Voigt and Löschel (2014) quantifies the economic effects of the Chinese ETS to be a GDP and welfare loss of 1% relative to BAU in 2020, with a corresponding carbon price of US\$7.46/tCO_2 in a medium growth scenario, and if allowances are auctioned. Simulations in Massetti and Tavoni (2012) indicated the benefits of an Asian ETS to China, which presents an importer of emissions allowances profile in the region. Qi *et al.* (2013) found that including China and the US, the largest carbon emitters, in an expanded ETS implies substantial impacts on the price and the quantity of permits traded internationally. Further, the literature has demonstrated that there are incentives for China to join a global ETS so as to avoid a carbon-based tariff imposed on its exports, which may not be the case for other developing economies which are less carbon-intensive and export-intensive (Hübler, 2012).

With the recent Chinese ETS, there are now 21 systems covering 28 jurisdictions in the world (ICAP, 2018). However, an increasing number of jurisdictions in developing countries are also planning, or at least exploring the potential for implementing an ETS, with major developments in Latin America. For example, a pilot Mexican ETS is scheduled to be initiated in 2018 and finish in 2021, when the system will be updated for the start of the formal phase, aligned with the start of the first period of commitment under the Paris Agreement. In fact, this is not the first carbon pricing initiative in Mexico. There has been a carbon tax of US$ 3.5 applied on fuel consumers since 2014, where the use of offset credits from domestic projects is allowed to comply with the tax liability. There is no specific sector targeted by the carbon tax, being much easier to regulate certain sectors through an ETS. In this sense, different climate policy instruments are expected to coexist in the future.

Mexico also envisions future linkages, particularly with markets in North America since it has signed cooperation via a Memorandum of Understanding

(MOU) with Québec. In addition, Mexico has been endorsing the use of carbon pricing as a key instrument for mitigation in the Americas, through the Paris Declaration on Carbon Pricing in the Americas of 2017. The initiative has the support of other developing countries such as Chile, Colombia and Costa Rica, but also Canada and the provinces of Alberta, British Columbia, Nova Scotia, Ontario and Quebec, along with subnational jurisdictions in the USA, namely California and Washington.

In parallel, there is dialogue taking place in terms of regional carbon pricing in the context of the Pacific Alliance, which is formed by Chile, Colombia, Mexico and Peru for exploring a voluntary market-based cooperation in Latin America (World Bank, Ecofys and Vivid Economics, 2017). For Colombia and Chile, integrating such a cooperative framework reflects the intention to implement an ETS in the medium term. In 2017, both have introduced carbon taxes and seek to link with other jurisdictions.

Several studies have investigated the effects of climate change mitigation in Latin America in terms of macroeconomic implications, abatement potential, financial investments and technological change in addition to adaption measures. In Feld and Galiani (2015), mitigation is only indicated to Latin America in the presence of a global binding agreement because it could allow the region to exploit existing comparative advantage. On the other hand, Clarke *et al.* (2016), Kober *et al.* (2014) and van der Zwaan, Calvin and Clarke (2016) suggest better responses from a constraint on emissions, if Latin America takes on less ambitious emission reductions. Associated costs to undertake these reductions via carbon tax vary from US\$15/tCO_2 to US\$50/tCO_2 by 2030. However, there is no indication on the use of an ETS to meet expected mitigation in the region in these investigations.

The ICAP (2018) report shows that another ETS scheduled to be implemented is the Ukraine ETS, whereas at the subnational level, the Nova Scotia ETS in Canada, the Taiwan ETS in China and the Virginia ETS in the USA have official dates to commence their market activities. According to this up-to-date document, many other jurisdictions both in developed and developing countries are considering the adoption of an ETS as part of their climate policy strategy, amongst them, Washington State and Oregon in the USA, Thailand, Turkey, Vietnam and Brazil. In the mid-term to long-term, and under the Paris Agreement framework, these systems could be linked to others.

To date, a small number of active national and subnational carbon markets are involved in, or are open to the concept of, ETS linkages[40]. Table 4 distinguishes the state of ETS links in already connected systems, planned links where negotiations have been concluded and are awaiting ratification to start operation, where a memorandum of understanding has been signed between jurisdictions intending to link in the future and finally, the possible linkages that have officially initiated discussions[41].

Table 4 – State of ETS linkages status in 2018

Existing linkages	Planned linkages	Memorandum of Understanding (MOU)	Potential linkages
• Ontario, California and Quebec • RGGI • EU ETS and Norway ETS	• EU ETS and Swiss ETS	• Mexico and North America (Quebec and Ontario)	• Virginia ETS (to be implemented) and RGGI • Oregon and WCI initiatives • Nova Scotia and Quebec-Ontario • NZ ETS and compatible markets (stated in the NDC) • Chile, Mexico, Colombia and Peru (Pacific Alliance)

Source: prepared by the author based on ICAP (2018).

The major example of an existing ETS linkage, and the largest in North America, is the California, Quebec and Ontario link. These jurisdictions are part of the Western Climate Initiative (WCI), a voluntary coalition of US states and

40 Linking ETS systems is the type of linkage mostly adopted, but not the exclusive.
41 It is worth mentioning that indirect linkages have also been proposed. For example, the California ETS in association with the States of Acre in Brazil and Chiapas in Mexico signed a MOU in 2010 to allow for the use of carbon credits from the REDD+ programme in those states to be compensated in the California ETS. However, it has not been initiated to date (Furtado, 2017).

Canadian provinces where there are common rules to facilitate cooperation for mitigation purposes. Reduction targets are set independently and participants are encouraged to harmonise regulations. Hence, the ETS elements are tightly aligned, while certain unique features still remain, such as the way offsets are used or revenues recycled (Purdon, Houle and Lachapelle, 2014).

The linkage was firstly established between California and Quebec in 2014, and Ontario joined the agreement in 2018. All ETS systems are very similar, particularly in terms of long-term emission reduction goals. For example, California, Quebec and Ontario committed to reduce 40%, 35.5% and 37% of emissions below 1990 levels, respectively. Similar scope also makes the ETS more compatible. This linkage also holds joint allowance auctions. An ex-ante study revealed that linking would reduce the carbon price and yield less leakage (Sawyer, Peters and Stiebert, 2016).

For Mehling, Metcalf and Stavins (2017), the relative similarity in carbon prices across partner jurisdictions in this linkage undercut the main economic benefits of linking, which is observed when marginal abatement costs are widely divergent. At the same time, this linked programme signals an unprecedented attempt to combine ambitious goals in a multilateral cooperation among jurisdictions, not sharing a common border or currency (Poloncarz, 2017). Not surprisingly, potential linkages may emerge with other jurisdictions, such as Oregon and Nova Scotia or Mexico via a MOU. Yet, the market-based strategy is only one element of the climate policy strategy in California, Quebec and Ontario, and not the most important one (Purdon, Houle and Lachapelle, 2014).

Another linkage included in Table 4 is the Regional Greenhouse Gas Initiative (RGGI) in the northeast of the USA. According to Ellerman and Buchner (2006), while this is not technically a link between independent ETS systems, because of similarities to linkage and the involvement of national or subnational jurisdictions in a cross-border market; it configures an example of multilateral linkage. The RGGI ETS programme aims at reducing emissions related to electricity production, thereby covering 21% of GHG emission only from fossil-fired power plants with a generation capacity above 25 MW within

Connecticut, Delaware, Maine, Maryland, Massachusetts, New Hampshire, New York, Rhode Island and Vermont (World Bank, 2014).

Likewise, the EU ETS, the largest and most consolidated system in the world, can also be characterised as multilateral linkage (Mehling, Metcalf and Stavins, 2017). In both cases, there is an explicit disposition to link with other compatible systems, which means other ETS systems with a similar environmental integrity and system architecture. There is a similar disposition from the NZ ETS and the EU ETS to agree on future linkages. In view of this, the yet to be implemented Virginia ETS has declared the interest to link together with the RGGI. The EU ETS has so far engaged in a link with the Norway ETS, and just recently finished negotiating a full integration with the Swiss-EU linkage, which has not yet been launched. A first attempt to form a transatlantic ETS had taken place when the EU ETS and a proposed Australian ETS discussed a bilateral international link, which had a setback after the Australian ETS was repealed in 2014.

Overall, the experience with ETS linkages is still limited. However, conditions are now favourable for expanding the reach of international cooperation and for introducing meaningful carbon prices across countries. The literature has assessed the implications of combining existing and emerging ETS systems using both theoretical and quantitative approaches. As the EU ETS legislation allows for linking at national or regional level, several potential candidates have been analysed, including those in developing countries. The establishment of a developed-developing country ETS linkage would serve as a model for other countries wishing to join an integrated ETS.

The impacts of linking the EU ETS to the US system were evaluated in Chapman (2009), Zetterberg (2012) and Marschinski, Flachsland and Jakob (2012). Authors found that linking slightly influences the level of emissions abatement, and competitiveness concerns are only partly addressed due to potential market distortions. Chapman (2009) focused on the harmonisation aspects of the linkage and suggested that integrating the EU ETS to the US ETS requires little or no harmonisation with respect to cost containment, allocation method, coverage, cap and price levels and offset use. In this context, there would be potential benefits from the linkage. Zetterberg (2012) investigates the prospects of linking the EU ETS to the California system in terms of relevant design features such as the cap and allocation method, concluding that political will would play a major role in overcoming barriers of linking them. Using a standard Ricardo-Viner model, Marschinski, Flachsland and Jakob (2012) decompose the welfare effect of linking the EU ETS and a hypothetical US ETS, into gains-from-trade and terms-of-trade contributions, and show that the latter can make the overall effect ambiguous.

Several studies have been carried out in order to assess the possibility of two-way linkage of the EU ETS with other non-EU schemes such as South Korea, China, Australia and California, or instead to a multilateral agreement. Hawkins and Jegou (2014) explore the design of an integrated system between South Korea and Europe as a mean to facilitate linkage. The study finds that South Korea tends to gain through the linkage, due to price convergence that would reduce its high carbon price, and, consequently, the compliance costs for covered entities. The EU ETS would benefit from this linkage proposal since it would turn out to be a net seller of allowances to South Korea. Jotzo and Betz (2009) evaluated the potential opportunities of the Australia-EU ETS linkage proposal and concluded that it would face a number of obstacles, in particular due to differences in environmental ambitions and the use of offsets.

Anger (2008) modelled a linkage with Canada, Japan and the Former Soviet Union and found that it leads to a strong fall in compliance costs of 60%, with a further benefit for non-energy intensive sectors if Australia and USA also participates. If credits originated from carbon abatement from reduced deforestation in developing countries are allowed, there is additional decrease in the post-Kyoto climate policy costs (Anger and Sathaye, 2008). In this case, tropical rainforest regions receive a substantial amount of revenues from exporting carbon-offset credits to developed countries, who also manage to tighten the carbon constraints without increasing mitigation costs.

Another study also employs an economic model to illustrate efficiency gains of emissions trading from the EU perspective, which underline the welfare costs of regulating only energy-intensive sectors (Klepper and Petterson, 2006).

Similarly, Marschinski, Flachsland and Jakob (2012) show that linking the EU ETS to a sectoral ETS system, the emerging Chinese ETS, and sharing a common coverage bears some negative implications for the EU, which could be addressed by linking across asymmetric ETS sectors, thereby being more acceptable for the EU. Moreover, the analysis indicates that a link between an absolute-based and intensity-based cap ETS system can generate leakage, through increased fossil fuel use in non-capped sectors, and could not be perceived as a facilitator for this link. Zeng, Weishaar and Couwenberg (2016) have a similar conclusion regarding the divergences of cap setting as a barrier to link.

The effects of sectoral ETS linkage, i.e. a linked framework where the emissions constraint is imposed on a limited number of sectors, are investigated under different circumstances. For instance, Gavard, Winchester and Paltsev (2016) modelled a sectoral ETS on electricity and energy-intensive industries in the EU, the US and China, simulating autarky and linkage scenarios. Hübler, Voigt and Löschel (2014) assessed a Chinese ETS regulating energy-intensive industries, electricity, heat, petroleum and coal products considering a potential cooperation with the EU ETS. Results from these studies showed an increased adoption of low carbon technologies, lower international leakage and generally, a greater degree of acceptance from developing countries to participate in the carbon market set by developed countries. Yet, this climate strategy involves small GDP and welfare benefits for China when linking to the EU ETS in Hübler, Voigt and Löschel (2014), even though China could potentially generate additional allowances for Europe due to the lower marginal abatement costs (Heindl and Voigt, 2012).

Similarly, Hamdi-Cherif, Guivarch and Quirion (2011) examine a sectoral ETS regulating only electricity generation in all developing countries and linked to developed countries ETS systems, which comprise all carbon emissions from fossil fuel combustion. Under this linkage approach, the authors found that economic impact in developing countries is milder than a global ETS since GDP losses and the effects on electricity prices are lower. Interestingly, results from Gavard *et al.* (2011) show that the ETS linking between the USA and China may yield only moderate increases in the generation of low-carbon energy whereas a rise in electricity price can be observed, with a negative impact on welfare. Additionally, the study highlights that the impact from linking ETS systems depends on the relative quantity of emissions in the two regions. For example, a linkage between the EU ETS and a hypothetical ETS in the United States has a larger impact on the EU carbon price than the linkage between the EU ETS or, alternatively a hypothetical ETS in Mexico or Brazil. In fact, under the modelling assumptions in Gavard *et al.* (2011), sectoral trading with those developing countries has little effect on the EU ETS emissions and carbon price, thereby implying smaller transfers of allowances to the EU.

Some empirical studies also consider the coordination of a multi-region ETS in which the EU ETS takes part. Dellink *et al.* (2014) use a global recursive-dynamic computable general equilibrium model to assess direct and indirect effect of linking ETS systems of developed countries, which leads to moderate cost savings due to the limited differences in permit prices in autarky. According to the study, the well-functioning of crediting mechanisms, even if used sparingly, are likely to create further economic benefits.

Qi *et al.* (2013) proposed a multilateral linkage among the EU, USA, Australia, New Zealand and China through the China-in-Global Energy (C-GEM) model in order to quantify implications on energy and emissions. Overall, China is the major net exporter of emissions permits and benefits from a renewable energy expansion. Whilst carbon prices are equalised to a lower level when China participates, the inverse effect is observed when introducing the USA to the integrated system.

Finally, in Xu *et al.* (2017) a conceivable multi-region ETS is simulated for China, USA, Europe, Australia, Japan and South Korea to identify impacts related to industry trade, the energy system, and international trade for each region. Because China is a large exporter of carbon permits, the study reveals that the abatement burden from the other regions is reduced, as well as the overall adverse effect on the economies with higher abatement costs, such as Japan and South Korea. Additionally, the link contributes to the development of clean energy in China while international trade is negatively affected.

In short, all the aforementioned investigations provide some common outcomes with interesting insights on linking to the EU ETS, for instance in relation to key features alignment. In this context, literature demonstrates that complete harmonisation is not fundamental for linkage to the EU ETS to succeed. However, evidence indicates that linking is not always beneficial for all participants, especially in the presence of market distortions (Flachsland, Marschinski and Edenhofer, 2009b).

From a review of literature on existing quantitative analysis, it is evident that there is a very limited number of studies aimed at estimating the potential opportunities for linking, particularly for developing countries in Latin America, where carbon pricing initiatives are becoming increasingly more common. To date, there has been little exploration of the effects of a developed-developing or developing-developing country sectoral ETS linkage.

In order to fill this gap, the framework introduced as part of this thesis considers linkage implications of a hypothetical Brazilian ETS with a similar sectoral coverage to the aforementioned studies. The implications of such proposals have to date not been investigated as carbon pricing and related linkages have just emerged as a reasonable alternative for developing countries. This is reflected by the late incorporation of climate issues into the Brazilian domestic agenda, that is, the secondary relevance given to environmental issues in light of other political national priorities. Additionally, it demonstrates that developing countries are envisioning financial opportunities from ETS systems. The expected benefits of accessing the market and joining a linkage are related to the exporter role developing countries would presumably assume (Somanathan, 2008). For a developing country such as Brazil, a climate alliance could be appropriate, if it promotes emissions mitigation simultaneously to economic development, in order to aid technological improvements and the transition to a low-carbon economy.

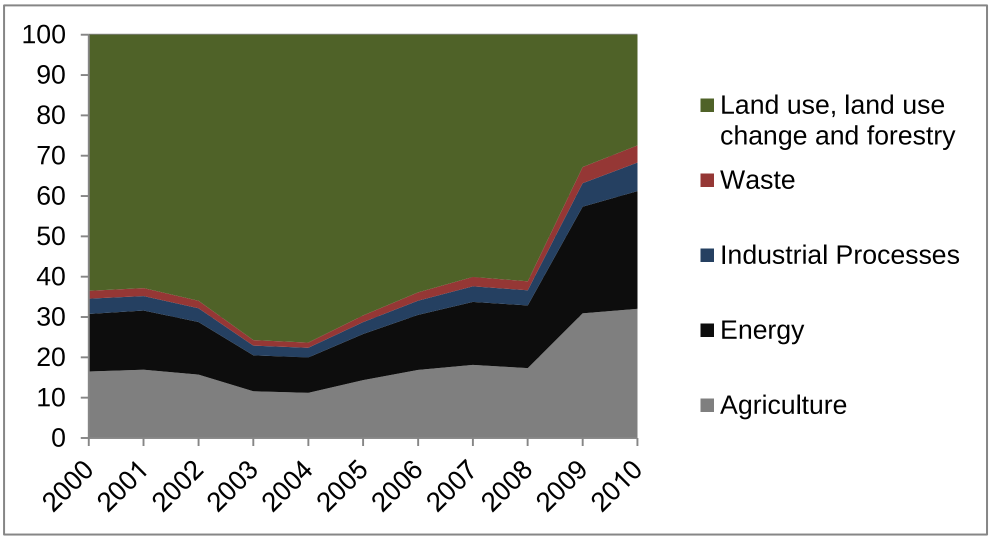
3.3 Climate policies and Emissions Trading Strategies in Brazil and Europe

3.3.1 Brazil

Among developing countries, Brazil has taken on a pioneering position when it comes to commitments to mitigate climate change. With approximately 4% of global emissions in 2014 (SEEG, 2016), Brazil had voluntarily agreed to reduce emissions during the Kyoto Protocol period, proposing to achieve the mitigation target of 36.1% to 38.9% by 2020, compared to emissions in 1990. Under the Paris Agreement, the country committed to promote a further cut of 37% and 43% of 2005 levels by 2025 and 2030 respectively. This is considered to be more stringent than the previous pledge, since it comprises an additional commitment to stop illegal deforestation.

Over the last decade, Brazil has greatly concentrated its efforts to control deforestation (Seroa da Motta, 2011) since it has been the major source of emissions in Brazil. By explicitly adopting national measures to reduce the rate of deforestation, the share of land use, land use change and forestry in total CO_2e emissions has dropped from 63.5% in 2000 to 27.5% in 2010, as depicted in Figure 3.

Figure 3 – CO2e* Emissions share (%) by sector in Brazil



* In terms of Global Warming Potential (GWP) values of 1995
Source: MCTI (2018).

Among those governmental initiatives one can mention efforts to prevent new deforestation, which includes the protected areas, thereby ensuring the enforcement of law (Nepstad *et al.*, 2014) as well as the sustainable management of native forest regions, such as the Amazon, in order to preserve ecosystems and biodiversity. To cope with that, a new institutional framework was introduced, the New Brazilian Forest Code (NBFC). In the NBFC, there are compensatory mechanisms for afforestation and land use recovery based on the use of carbon certificates, although it is still at an early stage. In fact, Brazil declared that strengthening and enforcing the implementation of the NBFC is perceived as one manner to help stop illegal deforestation by 2030.

From such downward trend in land use, land use change and forestry emissions, there has been a redirection of emissions composition, wherein emissions from agriculture gained momentum, especially from 2009. Compared to other countries in the world, the agriculture sector plays a major role in the Brazilian economy. With a future projection of growth in global food demand of 70% to 100% in 2050 (Godfray *et al.*, 2010), an increase in agricultural production in Brazil is likely in order to accommodate such demand. Accordingly, investing in low-carbon agriculture in Brazil is rather necessary.

Results from McKinsey & Company (2009) suggest greater abatement opportunities in controlling deforestation and emissions from agriculture. Recognising the relevance of agriculture and related-emissions, the Low Carbon Agriculture Plan (ABC Plan) was launched in 2009 as part of the National Policy for Climate Change (PNMC) for the purpose of mitigation, efficiency improvements and adaptation to climate change. The Brazilian NDC reinforces the relevance of this strategy for sustainable agriculture development. Yet, the study of Lima (2017) reveals that the main objectives of the ABC programme will not be achieved by 2020 if its application continues to be limited.

A similar upward trajectory is observed in the energy and industrial sectors. The share of energy emissions has grown from 8.8% in 2004 to 29.2% in 2010 while those from industrial process increased from 2.4% in 2004 to

7.1% in 2010. Final consumption of energy in industrial sectors is the greatest, corresponding to 32.9% (BEN, 2015), and emissions are generated through the burning of fossil fuels or from the process of transforming inputs. In combination with the transport sector[42], the emissions from the burning of fossil fuel in the industrial sector correspond to 72% of total emissions (Seroa da Motta, Couto and Castro, 2012).

According to an estimation by McKinsey & Company (2009), there is significant abatement potential in those sectors by 2030. The advantage for Brazil is to have a relatively low carbon intensity energy mix, a favourable position to gain competitive advantage in the context of transitioning to a low-carbon economy. Yet, the country still relies on the production and consumption of fossil fuels, which has the potential to hinder a genuine carbon mitigation towards sustainable levels. Therefore, climate policies aimed at energy-related sectors are required to help achieve national climate goals, as they correspond to approximately 36% of total emissions.

This is consistent with Ferreira Filho and Horridge (2017), who claim that reducing energy emissions is fundamental to help meet the Brazilian Paris commitments. In fact, Brazil has the intention to achieve 45% of renewables in the energy mix by 2030. This includes non-fossil fuel sources other than hydroelectricity. For example, the NDC projects an increase of at least 23% in the share of renewables in the power supply, particularly wind, biomass and solar. Also, it plans to expand consumption of biofuels to approximately 18% in the same period.

At the industry level, the only consideration made in the NDC is in support of promoting new standards of clean technology, energy efficiency and low carbon infrastructure. Part of the Brazilian strategy for tackling climate change, an equivalent sectoral strategy as the ABC programme on agriculture, has been introduced to consolidate a low carbon economy in the industrial sector, the so-called Industrial Plan (IP). The emphasis is on the largest energy-

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[42] The Brazilian NDC also proposed improvements in the transport sector, such as in the infrastructure for transport and public transportation in urban areas, as well as the adoption of additional efficiency measures.

intensive sectors, namely aluminium, lime, pig, iron, steel, cement, pulp and paper, chemical and glass.

The main objective of the IP is to prepare national industry to face a future of low carbon economy where carbon productivity is equally important compared to labour productivity, constituting a source of international competitiveness (MCTI, 2013). The design and implementation of the IP involves the gradual establishment of a MRV system on industrial emissions and an action plan to encourage abatement, which is subject to a 5% constraint compared to BAU emissions by 2020, when additional commitments will be evaluated. Rather than restrain economic development, the IP was set up to improve efficiency of industrial processes whilst preparing the sector to face the challenges posed by climate change issues. In comparison to other sectoral programmes regulated by the PNMC (Brazil National Plan on Climate Change)[43], there has been minor progress in terms of implementation (GVces, 2015).

Undoubtedly, considerable efforts are required to advance in promoting emissions reductions in those strategic sectors of the PNMC. Otherwise, the proposed voluntary commitment to cut emissions by 36.1% to 38.9% compared to BAU projections for 2020[44], or the Paris Agreement pledges for 2030, will be realised with the majority of contributions originated from controlling deforestation. This is due to the fact that abatement costs are lower in this sector than reducing emissions from agriculture or energy sectors, for instance (Gurgel and Paltsev, 2014).

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[43] Article 11 of the PNMC defined several sectoral plans to consolidate a low carbon economy. In addition to agriculture and industrial sectors, it implements strategies for transport and urban mobility, mining and health services. Those plans were elaborated with the involvement of representatives of sectors directly affected by the measures. Overall, every plan specified the emission reduction target for the 2020 period, actions for implementation, as well as relevant indicators to monitor and evaluate the effectiveness based on sectoral estimations of costs and impacts.

[44] This target was firstly communicated at COP-15 in Copenhagen, where Brazil made commitments that were afterwards confirmed as a national climate objective. It represented an important progress towards establishing a legal framework to regulate actions for mitigation and adaptation in Brazil, but also because it underlined Brazil's role in the international negotiations compared to other developing countries (Seroa da Motta, 2011).

The Brazilian literature has so far mainly evaluated the stringency and achievability of pledges under international commitments on climate policy without international cooperation. The modelling exercises consider the economic and environmental characteristics of Brazil and provide recommendations for mitigation at lower cost taking into account macroeconomic and distributional implications.

Using the EPPA model, Gurgel (2012) quantified the effects of a global transition to a low carbon economy on the Brazilian economy and emissions from 2020-2050. The author implements sectoral targets and a sectoral ETS in order to accommodate the different mitigation strategies proposed for each sector. Results indicate substantial GDP losses in 2050, which could be reduced if Brazil participates in a global ETS or instead, if trade tariffs based on carbon content were imposed to compensate for a hypothetical inexistence of climate policy in Brazil. There are large negative impacts on the energy sector and transport, with smaller impacts in the energy intensive and agricultural sectors.

Similar research was undertaken by Gurgel and Paltsev (2014) to analyse the alternatives available for Brazil to meet the climate compromise of COP-15 (and PNMC). Among the scenarios, market-based instruments are considered. One of them applies a carbon tax in each sector of the economy, in another the carbon tax is imposed only on emissions from deforestation. An ETS is also modelled covering energy use and agriculture, while emissions from deforestation are under the carbon tax in the same scenario, as opposed to the economy-wide ETS scenario. When a carbon tax is implemented, results reveal a very costly strategy of approximately US$290 per tonne of CO_2e in 2030, with higher prices in energy-intensive industries, services and transport. The ETS for all sectors implies a lower carbon price of approximately US$100 per tonne of CO_2e, where energy and agriculture sectors benefit from this. Although the adoption of an ETS regulating all sectors would be the best option, the authors recommend focusing on decreasing deforestation, and ensuring improvement in methods to measure emissions in the industrial and agricultural sectors, in order to make their inclusion possible in any future ETS.

A previous study of Feijó and Porto Jr. (2009) using GTAP-E has come to a common conclusion with respect to alternatives for Brazil, i.e. participating in emissions trading set in the Kyoto Protocol offers the best option as the country would have welfare gains. At the same time, it corroborates with the hypothesis that climate policies negatively affect the economy. The simulation with IMACLIM-BR of Wills and Lefevre (2012), for example, indicates that the sectoral cost of a tonne of CO_2 yields a GDP loss from 1% to 4% depending on the way revenues are recycled. As a consequence, there is a higher rate of unemployment than in BAU projections. However, in the presence of mitigation targets for Europe and the USA, macroeconomic impacts on Brazil are negligible (França and Gurgel, 2018).

A sectoral carbon tax is applied on emissions from industry and energy sectors in Lucena *et al.* (2014). The study compares results of six different energy-economic or integrated assessment models under different scenarios for carbon taxes and abatement targets up to 2050. The carbon taxes range between 32 US$/t$CO_2$e to 162US$/tCO_2e in the 2020-2050 period and induce emissions reductions due to lower energy consumption, increased penetration of renewable energy (especially biomass and wind), and carbon capture and storage technologies for fossil and/or biomass fuels. According to results, a combination of instruments is required for mitigation in Brazil, but also investments in research, development and demonstration (RD&D) in order to make technologies not yet technically mature in the energy sector available, as well as incorporating actions for abatement in agriculture, forestry and land use sectors.

The achievability of the Brazilian pledges under the Paris Agreement has been investigated in Carvalho *et al.* (2018), who estimated potential impacts of such a commitment by employing the BeGreen model. As expected, a decrease in GDP is observed, and the sectors with the greatest dependence on the burning of fuels, or with intensive emissions in their production processes, were the most negatively affected.

Some other investigations quantitatively evaluated the impacts of a Brazilian ETS regulating energy-intensive sectors, and found a similar decrease in macroeconomic indicators. Domingues, Magalhães and Carvalho (2014) employed the BeGreen model to estimate the feasibility and costs of imposing an emissions constraint via an ETS on industrial sectors by 2030. Emission reductions are set at 5% and 10% in 2020 and 2030, respectively. Results demonstrated that joining a sectoral ETS leads to GDP losses, but at a lesser magnitude than when there is no emissions trading allowed. The ETS would prevent the loss of R$270 billion between 2016 and 2030. A similar level of emission reduction is observed in both policy cases, with a corresponding carbon price of US$ 146[45] per tonne of CO_2e in 2030. Yet, there is a carbon leakage effect to the agriculture and energy supply sectors. A major conclusion is that a cap-and-trade mechanism in Brazil would be more cost-effective than a simple command-and-control regulation, as advocated by theoretical literature.

With a sectoral ETS regulating domestic energy-intensive industries, Rathmann (2012) assesses the policy response in terms of competitiveness effects. From the simulation using a hybrid model (Input-Output plus energy model), the loss of competitiveness is mostly verified in the oil refining and steel (ferroalloys, non-ferrous metallurgy and pig iron) industries, requiring the adoption of compensatory mechanisms. To cope with that, allocating carbon permits for free and encouraging the adoption of low carbon technologies are fundamental strategies in the short term and long term. For the petroleum refining segment, a different mitigation target is suggested.

Distributional implications of setting up a sectoral ETS configure the main objective of (Castro, 2013). In order to measure efficiency and equity of two different allocation criteria within the industrial sector, simulations are based on estimations of the Marginal Abatement Cost (MAC) curve[46]. Pricing the carbon reduces abatement costs by 78-82%, but depending on the criteria used to allocate emissions allowances, different costs and revenues are observed. The carbon price found for a target of 30% was US$301/t CO_2e, with a net revenue

[45] This is equivalent to R$388 but converted according to the exchange rate R$/US$ of 2014 (2.66) for comparability purposes.

[46] Under the climate perspective, MAC curves are an important element to indicate where emissions abatement opportunities are, and given the abatement technology available, how much emissions can be avoided by a sector.

corresponding to US$76 billion, which is reduced to US$ 52 billion if a more protectionist scenario takes place.

Similarly, Grottera (2013) uses a Social Accounting Matrix (SAM) for the Brazilian economy in 2005 to investigate the effects of carbon pricing on income distribution. Again, depending on the carbon price (R$25 or R$50) and how the revenue is recycled into the economy (lump-sum transfer to low income families or exemption from labour taxes), income distribution differs, as do the macroeconomic effects. In Carvalho *et al.* (2018), putting a carbon price on economic sectors to comply with the Paris Agreement pledges has a negative effect on household consumption, with the lower income being relatively more affected due to the price increase of agriculture products and food.

In general, although resulting in some significant implications for the economy and income distribution, emissions reductions can be achieved cost-effectively with a domestic ETS. From those studies, there also seems to be a consensus about the advantages of participating in a global, or at least a wider scheme, in order to pursue greater cost savings. Ultimately, although little consideration on the overall ETS design is made, those empirical evidences corroborate with theoretical literature on cap-and-trade schemes indicating the cost effectiveness of an ETS in Brazil.

However, contrary to some other countries, the Brazilian NDC does not specify the use of market-based mechanisms to comply with national pledges for 2020-2030. In recent years, the debate on adhering to carbon pricing strategies has evolved in Brazil, but at a slow pace. The enactment of the PNMC in 2009 was fundamental in this respect since it envisages financial and economic instruments at domestic level as a means to achieve emissions mitigation and promote adaptation by 2020. Note that although no principles for emissions trading have been included, it forms the basis for a future ETS. The only involvement Brazil has in carbon market mechanisms is via the CDM projects of the Kyoto Protocol, through the platform for negotiating carbon credits in the stock exchange (BM&F), the so-called Brazilian Market of Emissions Reductions (MBRE)[47].

Since 2014, more than 20 companies of different economic sectors have been participating in a pioneering ETS simulation in Brazil (and in Latin America). This voluntary initiative intends to disseminate knowledge on the functioning of the ETS, to prepare the industry sector to ensure it has capacity to integrate a domestic system and ultimately, to develop proposals for a robust Brazilian ETS. In the platform, allocation and trading is managed by the Green Stock Exchange of Rio de Janeiro (BVRio[48]) whereas the elements for the ETS design are coordinated by the Centre for Sustainability Studies of Getulio Vargas Foundation (GVCes/FGV). An important lesson from this exercise is that, for a hypothetical ETS in Brazil to succeed, there must be substantial progress on the MRV system in addition to technical and administrative capacity to operate the ETS (Nicolletti and Lefévre, 2016). In 2012, a state-wide ETS has been considered for Rio de Janeiro and Sao Paulo, which was afterwards aborted (ICAP, 2018).

The Brazilian government has been supporting, in association with the World Bank - Partnership for Market Readiness (PMR), a comprehensive group of studies based on carbon pricing for the post-2020 period[49]. The aim is to bring lessons from existing initiatives that could be applied to a similar system in Brazil, whilst evaluating requirements and potential implications of market-based instruments for domestic mitigation. A particular policy package proposal based on the use of carbon tax and ETS is under evaluation, which could become a White Paper with design recommendations for Brazil (ICAP, 2018).

Despite that, Brazil has not yet defined or even decided on whether to implement a domestic ETS. However, the arrangements for market instruments in the Paris Agreement may encourage Brazil to design a carbon trading

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[47] Operations in the MBRE are foreseen under the PNMC. The existence of such a structure is fundamental for possible trading of emissions, at both domestic and international level in the future.

[48] BVRio promotes the use of market-based instruments to facilitate the enforcement of environmental laws as well as to support green economy in Brazil. There is a carbon market platform for companies to negotiate and trade environmental assets in the form of allowances, offsets, and other carbon-linked financial products.

[49] See more information at: https://www.thepmr.org/country/brazil-0

system. By taking the lead, Brazil may encounter new opportunities for climate cooperation with developed systems, with the EU ETS being a potential candidate, or other emerging schemes in developing countries such as in Latin America, or the Chinese ETS. On the basis of common sustainable development priorities, Brazil recognised the role of South-South cooperation in some areas, for instance, forest monitoring systems, biofuels capacity-building and technology transfer for low-carbon and resilient agriculture. In light of that, accepting international cooperation via ETS linkage could also be envisaged in the future.

Due to the complexity associated with implementing a bilateral linkage as part of the climate architecture, addressing climate change requires cooperation whereby regulatory and market mechanisms are considered appropriately. Domestic use of carbon pricing instruments still faces political and economic opposition in Brazil, as well as in other emerging economies. Therefore, international coordination has an appealing virtue of providing efficiency gains along with significant emissions reductions that could compensate the sector-specific costs of an ETS, ultimately giving rise to political acceptability. Harmonisation may have the potential to avoid distributional effects and help to effectively provide environmental and economic benefits. The focus of this investigation is to understand whether a combined ETS between Brazil and Europe would, thus, satisfy the main objectives of the climate policy.

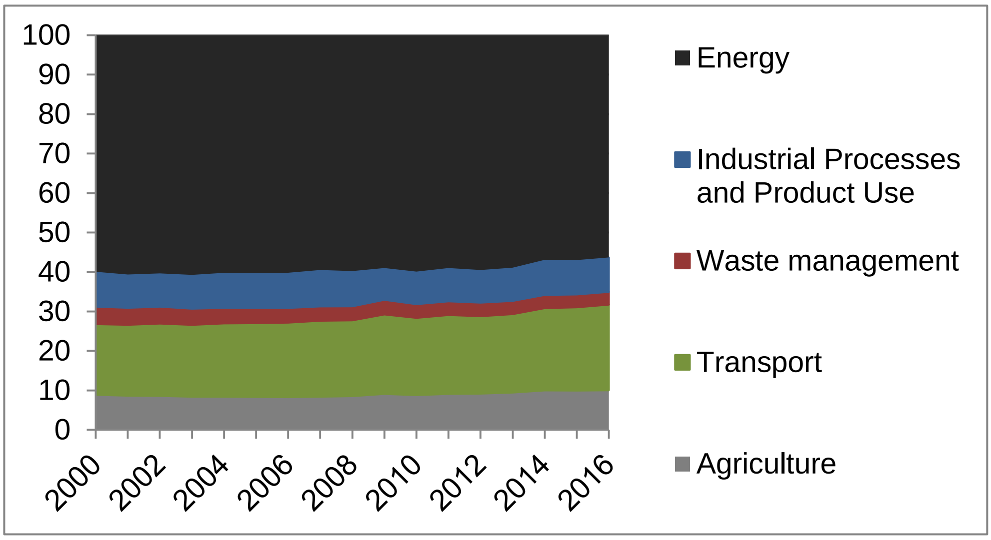
3.3.2. Europe

Over the last decades, the EU has increasingly invested efforts in the decarbonisation of the economy, which is a reflection of both domestic commitments and those agreed in international climate forums. In order to progress in achieving this long-term objective, a combination of climate strategies has been adopted with the aim of taking action in several areas. This includes regulatory and economic instruments for policy enforcement. The observed role of carbon pricing in the EU, and lessons from its adoption, are useful to orientate emerging market-based approaches and related-linkages.

Given the emphasis on environmental outcomes, the EU strategies translate the objective of promoting efficient mitigation, and investments in low carbon technologies (Laing *et al.*, 2013). At first, the EU introduced the climate energy package comprising initiatives for increasing energy efficiency and the use of renewable energy to ensure a 20% GHG emission reduction by 2020 compared to 1990 levels. In line with these, the EU agreed on the emissions and energy path for 2030, which define a commitment of at least 40% GHG emission reduction at domestic level compared to 1990 levels (European Commission, 2013). Additionally, the EU also set up a roadmap for transitioning into a more competitive and low-carbon economy in 2050, particularly addressing energy and transport.

These are the major sources of emissions in Europe, accounting for approximately 80% of total CO_2e emissions from 2000 to 2016, as displayed in Figure 4. The energy sector has undoubtedly the largest share of total emissions, demonstrating that despite substantial efforts to reduce fossil fuel use, further decarbonisation is still required. According to Dechezleprêtre, Martin and Bassi (2016), for a full decarbonisation in the EU power sector, the share of low-carbon sources must increase so as to compete with fossil fuel-based power, particularly in the presence of a carbon price signals.

Figure 4 - CO_2e* Emissions share (%) by sector in Europe



* In terms of GWP values of 1995
Source: EEA (2018).

The major strategy to help drive emission reductions in the European energy sector is the EU ETS, which also plays an important role in cutting emissions from large industrial installations. Emissions from industrial process and product use correspond to approximately 10% of the regions CO_2e emission, and have remained relatively unchanged between 2000 and 2016. Together, 45% of the EU emissions are comprised in the ETS, encompassing over 11,500 installations[50]. For effectively incentivising these sectors, as well as the aviation sector, to mitigate emissions at minimum cost, the EU ETS set up a 21% and 43% cut in emissions by 2020 and 2030, respectively.

The EU ETS was designed as an element to provide flexibility for the EU and Member States to achieve the targets of the Kyoto Protocol. It was formally systematised by the Emissions Trading Directive in 2003, and subsequently, launched in 2005[51], being the first experience with a nationwide ETS in the world (Ellerman, 2009). After ten years of operation, the EU ETS offers

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[50] Liable entities in the EU ETS are defined at the installation level and the level of aircraft operator.
[51] A “Linking Directive” followed the Emissions Trading Directive and integrated the EU ETS and the Kyoto Protocol’s flexibility mechanisms. Those installations participating in the EU ETS and obtaining credits from emission reduction projects in the Kyoto Protocol would be allowed to use them in a certain proportion to fulfil remaining obligations under the EU ETS.

important lessons for other regions envisioning a low-carbon economy in terms of designing an ETS.

The EU ETS operates by setting an absolute limit on covered emissions and issuing a number of tradable permits - also known as European Emissions Allowances (EUAs) - that is equal to the cap. As the EU ETS is divided into phases of compliance, different rules have been applied to the cap setting and for other ETS design features. For example, in the first two trading periods (2005-2012) the cap was set at national level based on National Allocation Plans (NAPs) of every Member State, whereas from the third phase the cap was implemented with gradual increase in the level of stringency (1.74% and 2.2% annual decrease in the cap for the third and fourth trading period). This is a peculiarity of the EU ETS, that is, to set an ETS cap in a context of another cap, the targets of the EU climate policy (Ellerman and Joskow, 2008). At the same time, coordinating such governance structure has been very complex, ultimately giving rise to doubts about the fairness of the ETS (Zetterberg *et al.*, 2012).

Similarly, the method of allocating emissions permit in the EU ETS has evolved over trading periods. Each allowance gives the right to emit one tonne of CO_2 equivalent and each covered installation surrenders a number of allowances equal to the verified CO_2 emissions in the year, but can opt to sell or save the unused ones for future periods. In the first two phases, free permits were allocated up to 95% and 90%, respectively according to historical emissions. Particularly in the first phase (2005-2007), when there was the intention to acquire experience through a learning-by-doing process, a cost-free allocation was adopted for attracting industry acceptance, to which economic costs from auctioning could be detrimental. Although Tietenberg (2003) asserts that allocating emissions permits cost-effectively independent of initial distribution, the transitory aspect of grandfathering is widely accepted worldwide, since most of ETS systems adopt it in pilot periods.

The EU ETS also uses free allocation to protect certain industrial sectors from the risks of carbon leakage or exposure to international competition. This is a major source of concern from the regulated entities perspective, but so far empirical studies have not found the negative impacts on competition for regulated firms (Laing *et al.,* 2013). From the beginning of the third phase, 80% of industry allowances were still allocated for free, whose proportion is projected to gradually reach 30% by 2020. The aviation sector also surrenders the majority of allowances for free. In contrast, the electricity sector has started to buy allowances via auctioning. This shift occurred because the capacity to pass through carbon-related costs to consumers was observed, even though there are some cases where those firms with surpluses have opted to not pass through the savings (Ellerman, Convery and Perthuis, 2010). Yet, Laing *et al.* (2013) report that some industrial sectors have been passing through costs as well. From the consumers' perspective, empirical studies show tangible effects on households due to the rise in electricity prices, implying distributional effects (ZEW, 2016).

One important benefit of free distribution is that it enables firms which would not have been capable of purchasing permits to remain in business. On the other hand, it allows those entities able to pay for allowances, or make internal abatement decisions, to get them with no cost (Egenhofer, 2007). From the EU ETS experience point of view, grandfathering has demonstrated the potential to create an extra benefit, i.e. windfall profits, encouraging regulated parties to engage in rent-seeking behaviour in order to obtain more generous future allocations of allowances (Zetterberg *et al.*, 2012). Further, it created a wealth transfer to companies granted with free allowances, but with no implications on competitiveness or emissions leakage (Schmalensee and Stavins, 2017a).

Not surprisingly, the EU ETS had a significant surplus of allowances during the period, more specifically 83 million allowances at the end of phase I and 1.8 billion allowances at the end of phase II (European Commision, 2015). As a result, over-allocation has led to a collapse of the carbon price (Ellerman, Marcantonini and Zaklan, 2016), less than US$10 per tonne of CO_2 (Rydge, 2015). Literature suggests that such price crash, and the continuous low level of emissions permit price since 2008 has been a reflection of the economic recession and renewables-promoting policies, which provoked a drop in the allowances demand (Doda, Taschini and Druce, 2017) but also the inability of the ETS to respond to changes in economic circumstances (Grosjean, Acworth and Flachsland, 2014). This is considered to be a low emissions price for stimulating the adoption of environmentally friendly technologies and innovation in low-carbon technologies (Dechezleprêtre, Martin and Bassi, 2016) and ultimately, to promote an efficient control over emissions (Nordhaus, 2008).

In terms of emissions reductions, the EU ETS has been a cost-effective instrument for mitigation, despite the functioning problems (Doda, Taschini and Druce, 2017). Some studies underline the environmental effectiveness of the EU ETS (Ellerman, Convery and Perthuis, 2010; Anderson and Di Maria, 2011). According to Laing *et al.* (2013), evidence suggests that the largest emission reductions in the power sector were due to the switch from coal to gas-based generation, while abatement has also been achieved in the cement sector as a result of substitution to alternative energy sources or decreases in the clinker content (Ellerman, Convery and Perthuis, 2010). For the 2030 period, there are high stakes for the decarbonisation of the ETS sectors in light of carbon intensive technologies and infrastructure phasing out, with potential to help assuring energy security in the EU (Sartor *et al.*, 2015). However, if complementary policies for enhancing energy efficiency or the share of renewable are combined, potential inefficiencies may arise (Tvinnereim and Mehling, 2018).

Hence, the EU ETS must address technical challenges. One key message is to prevent unnecessary price fluctuations and over-allocation in order to guarantee the well-functioning of the ETS, as well as the benefits of the carbon market in tackling climate change. To cope with that, the European Commission (EC) has been attempting to strengthen the EU ETS by revising the rules and proposing amendments so as to enhance the stability of price signals and to further induce innovation and the use of low-carbon technologies. Currently in Phase III (2013-2020), the EU ETS has broadened the scope and created a Market Stability Reserve (MSR) in an attempt to offer a stronger incentive to reduce emissions and to improve the functioning of the carbon market by neutralising negative effects of allowance surpluses and hence, resilience to future shocks from 2019 (European Commission, 2015a). There is also potential co-befits of auctioning, as 50% of revenues generated are set to be used for climate and energy related purposes. From the €3.6 billion of revenues in 2013, €3 billion will be invested in energy efficiency, renewables, research and sustainable transport (European Commission, 2018).

Considering an appropriate approach to the decarbonisation of non-ETS sectors is also needed. Emissions from non-ETS sectors are subject to a reduction target of 30% compared to 2005 levels by 2030. However, a recent report of the European Environmental Agency (EEA, 2016) shows that is likely to fail in meeting these targets. Bassi *et al.* (2017) suggest imposing a carbon tax on these emitters as it is relatively simple and would force them to manage the carbon output. In the future, this could be accommodated in the ETS.

Another important strategy of the EU ETS for future trading periods is the engagement in international cooperation via carbon markets, as already highlighted. In this case, whether or not to link the ETS depends on the partner to accept some established conditions[52], ultimately associated to the sharing of climate goals, which can be translated into ETS caps coherent to the global long-term commitment to limit the temperature increase to 2°C. Linking to a less ambitious partner could be conceivable for strategic reasons; however the EU tends to prefer cooperation partners with comparable ambitions, climate policies, and similar medium to long-term emissions trends (Flachsland *et al.*, 2008b). Since these elements reflect environmental ambition and aggregate goals for the integrated system, improper alignment or unequal decisions may interfere in the environmental effectiveness of the policy (Burtraw *et al.*, 2013; Kachi *et al.*, 2015).

For this reason, Hawkins and Jegou (2014) argue that trade-offs that might undermine the integrity of the linked ETS system would give rise to more concern for the EU than an increased carbon price, for example. This is not applied for other ETS design elements, which need to be technically negotiated. This was the case with the Swiss link to the EU, whose main point of

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[52] See EU official online documents available at: https://ec.europa.eu/clima/policies/ets_en.

disagreement in this link was the inclusion of the aviation sector. Differences from the Norway ETS have also been aligned prior to the link with the EU ETS. These two cases illustrate that linking to the EU ETS may require a degree of harmonisation and demonstrate that usually amendments are made from the partner side. Therefore, it is rather unlikely that the EU would accept linking to an independently-designed ETS.

Since there are barriers to terminate the link (Pizer and Yates, 2015), one crucial consideration for each jurisdiction is to evaluate if the benefits of linking outweigh the sovereignty loss over decisions on the ETS design (Green, Sterner and Wagner, 2014). All those aspects are highly relevant and must be carefully negotiated for developing countries aiming to design an ETS, and link it to existing developed-world programmes such as the EU ETS. Policymakers need to be aware of the political feasibility of such a proposal, as well as the economic and environmental implications and hence, the appropriateness of the strategy to help reducing emissions. This modelling exercise contributes to the literature by providing this evidence in the context of a proposed Brazilian ETS linking with the EU ETS. The methodology underpinning the modelling exercise is discussed in Chapter 4.

CHAPTER 4
METHODOLOGY: ECONOMIC MODELLING USING EPPA6

4.1 Introduction

The use of quantitative exercises to investigate economy-wide impacts of mitigation policies is fundamental. They can help determine and formulate the most feasible strategy whilst attempting to anticipate potential negative outcomes, thereby exerting a powerful influence on climate policy-making decisions. There are mainly two modelling approaches available for energy-related policy simulations, bottom-up energy system models and top-down models of the broader economy. Bottom-up models display a great focus on technological details, being able to capture the least-cost combination of energy system activities in the presence of technical restrictions and energy policy constraints (Böhringer and Rutherford, 2005). In contrast, top-down approaches have the advantage of providing a more comprehensive representation of the economy interactions and thus, to evaluate economy-wide feedbacks from market adjustments given a policy shock[53].

The majority of empirical literature from Chapter 3 applies top-down models to capture economy-environmental interactions and energy related problems, particularly Computable General Equilibrium (CGE) models[54]. The increasing popularity of CGE modelling over other methods of carbon policy analysis stems from its ability to translate policy shifts into economic responses of agents across multiple sectors and regions (Balistreri *et al.*, 2015). The CGE approach is useful for such analysis since general equilibrium effects on the

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[53] Typically, one important disadvantage of top-down models lies in the lack of details on current and future technological alternatives, which may be an important aspect for energy policy proposals (Andersen and Termansen, 2013). As an alternative, there has been a significant effort to combine the strengths of top-down and bottom-up approaches into a hybrid framework that could compensate their limitations.

[54] These are macroeconomic and Input-Output (IO) models. CGE models are an evolvement of IO models and linear programming (Ferreira Filho, 2011). In the context of climate policies, econometric models are not commonly adopted since estimates of future climate policy impacts depend on observed information, which not always is available. Further, they are not able to capture the effects of energy policies on the allocation of resources.

resource allocation of economies can be determined from the implementation of carbon constraints.

Additionally, CGE models are usually preferred because of the explicit treatment of economic behaviour, which is based on microeconomic optimisation assumptions (Zhang and Folmer, 1998). In CGE models, each economy is represented through a set of producers, consumers and governments, which are interconnected by markets for commodities and factors along with taxes and subsidies (Wing, 2011). At the producer side, decisions are made rationally in order to maximise profit under constant return of scale, whereas firms act to maximise utility. There are similar optimisation assumptions to describe government's behaviour.

In the context of climate policies, changes in relative prices and quantities, as well as the behaviour of producers and consumers, leads to a shift away from high-carbon energy sources, implying economy-wide effects from the carbon constraint imposed. As discussed in Clarke *et al.* (2016), the level of decarbonisation in an economy is different among CGE models as well as the costs. For example, models with broader low-carbon options generate less costly effects and those models built on myopic decision making (recursive dynamic) generally have higher economic costs. International trade representation and assumptions on capital mobility also have implications for the cost of carbon policies in the CGE model.

Theoretically, CGE models consist in an application of the Walrasian general equilibrium system (Walras, 1969), or an algebraic formalisation of the abstract Arrow-Debreu general equilibrium structure (Arrow and Debreu, 1954), in which equalisation of supply and demand in all interconnected markets is possible and occurs through changes in relative prices, which are determined endogenously. In other words, as a result of supply and demand decisions, there are price adjustments of commodities and production factors to guarantee equilibrium across the set of markets[55]. In light of the considerable time it can

[55] As a matter of model consistency, aggregate flows of the economy also need to be in equilibrium. For this reason, the macroeconomic closure of the model is defined, which in general reflects a certain macroeconomic theoretical approach (Ferreira Filho, 2011).

take for prices to adjust, CGE models are essentially conceptualised as long-run models (Fankhauser and McCoy, 1995).

A CGE model has functional forms and parameter specifications that translate the economic functioning as well as incorporating theoretical and empirical information. For instance, the model uses parameters in utility and production functions to represent tastes and technologies and capture substitution possibilities. For calibration purposes, CGE models use real-world databases as numerical references, usually input-output accounts which are supplemented by estimations of elasticities parameters. Numerically, the general equilibrium problem is solved via computable mechanisms, such as the General Algebraic Modelling System (GAMS), which is an optimisation programme (Brooke *et al.*, 1998). The equilibrium solution of a CGE simulation provides very detailed results on market clearing prices, sectoral output, commodity and factor substitution, income and economic welfare and GHG emissions, etc.

However, there are inherent limitations of CGE models for practical policy decisions. Since those models are grounded in microeconomic theory, a major criticism is related to the capacity to represent real world events, given that there are cases where producers and consumers do not behave in line with theoretical assumptions, and in many circumstances market failures exist. More importantly, the hypothesis that transactions occur only when the economy is in equilibrium is considered to be improper, especially because it suggests that economic decisions are made from the equilibrium stage. Another argument is the lack of empirical validation with historical data in the calibration procedure (Zhang and Folmer, 1998). The model structure is highly complex and it is calibrated through several parameters, estimated independently from existing studies, rather than under a general-equilibrium approach. Overall, results of the climate policy in CGE modelling are directly influenced by the assumptions, including those related to alternative technologies, and definition of the reference scenario, ultimately being highly sensitive to changes in parameter values. Accordingly, it is recommended to extensively investigate the sensitivity of results for a more robust analysis.

In fact, rather than providing a precise numerical response, the relevance of observed results to policy recommendations is to indicate long-run tendencies and the approximate magnitude of impacts from exogenous shocks in the economy, and further to compare it with alternative scenarios. This is the purpose of this research. To develop the ex-ante analysis on the effects for Brazil of adopting a climate policy via a sectoral ETS, and thereafter, a link with the EU ETS or alternative partners in the period 2020-2050, we use EPPA6[56]. Economic modelling is necessary as most of the economic indicators of such an ETS linkage cannot be quantified empirically. In this chapter, we present the main characteristics of the EPPA6 model and define the modelled scenarios under investigation in this thesis.

[56] Compared to previous versions, EPPA6 has been improved or updated in order to better represent some observed economic features, which includes the revision of the database and benchmark year, changes in the capital vintaging structure of the model or in the nature of economy-wide productivity.

4.2 Characteristics of the EPPA model

EPPA6 is a dynamic-recursive CGE model developed by the MIT Joint Program on the Science and Policy of Global Change and has been widely applied to energy and climate-related analysis. The model is written as a nonlinear complementarity problem in the GAMS programming language (Brooke *et al.*, 1998), using the syntax of the MPSGE (Mathematical Programming System for General Equilibrium) algorithm developed by (Rutherford, 1999).

As a CGE model, EPPA6 can represent the global production and consumption of various sectors of each regional economy and the associated GHG emissions, being interconnected to other regions through international trade. EPPA6 is solved for a sequence of global market equilibrium considering "myopic" expectations of economic actors that provides a representation of the global economy (Chen *et al.*, 2015). This assumption in EPPA means that current period investment, savings, and consumption decisions are made on the basis of prices in each 5 year period (Reilly *et al.*, 2007). The model considers a long run simulation horizon (2010-2100), being solved at 5 yearly intervals.

In each period, there are production functions in all sectors that describe the use of primary factors (capital, labour, and energy resources) and energy and intermediate inputs for producing goods and services for every region or country. The general rule for exhaustible resources, such as fossil fuels, is to have a decrease in the stock to the extent they are used, resulting in higher costs of extraction. The higher the output and income levels, the greater the demand for goods produced by each sector. This ultimately leads to higher production costs, as these goods use finite natural resources in the production cycle.

Similar to other CGE models, EPPA6 is designed using a nested Constant Elasticity of Substitution (CES) structure[57] to describe consumers'

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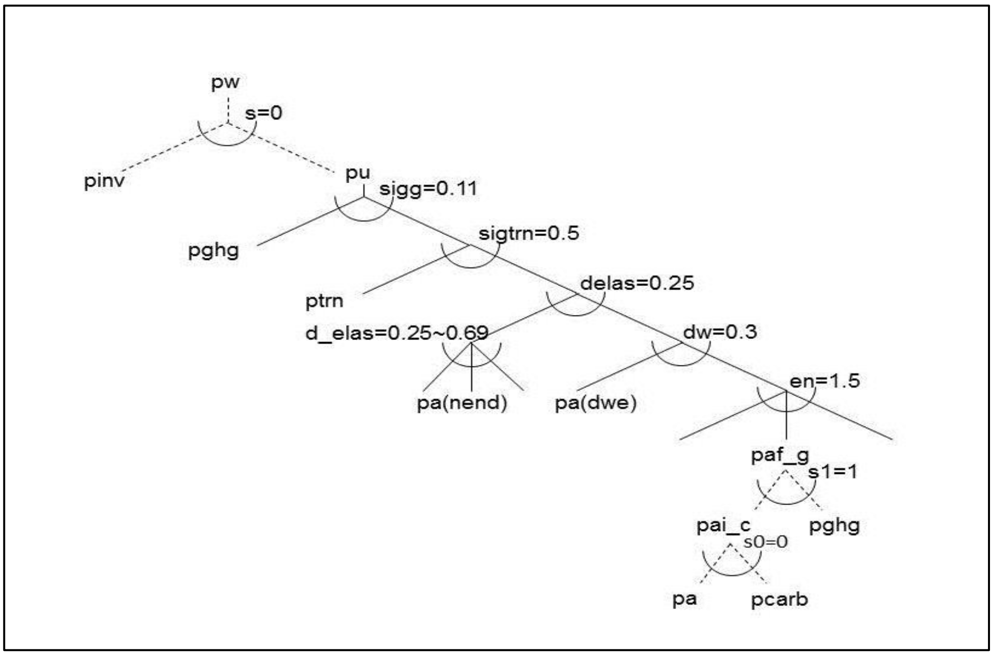
[57] These are functions with constant return of scale, which means that doubling inputs lead to a double output. Leontieff and Cobb-Douglas functions are CES particularities also adopted in EPPA6 representation. According to Paltsev *et al.* (2005), the use of nesting structure provides flexibility in setting elasticities of substitution, notably those related to fuels and electricity, or elasticities to which emission and abatement costs are particularly sensitive.

preferences among goods and the willingness to substitute them for others due to changes in relative prices; and production technologies, which determine how much of intermediate and primary factors are necessary for the production of a unit of output and the ability to substitute among available inputs. The rate of substitution is determined by elasticities. The government is a passive entity, which finances government consumption and transfers with revenue from taxes paid by households and producers. Deficits and surpluses generate return to consumers as lump sum transfers.

The model considers, therefore, three types of agents in each economy, that is, household, producers and government. To represent the consumption sector, there is a representative agent who is in control of primary factors and seeks to maximise utility by choosing how to allocate income received from the services provided (wages, capital earnings and resource rents) across consumption and savings.

Figure 5 – Nested structure of the utility function*



* Dashed line denotes a separate function
Source: Chen *et al.* (2015).

The setting to represent the utility function for the household sector is presented in Figure 5. Since savings enter the utility function, consumption-investment decisions are made endogenous and thereby allows for the capacity of future production and future consumption levels to increase. Yet, in order to avoid double counting of changes in savings over time, the welfare accounting in EPPA6 is reported in terms of changes in aggregate consumption, where results for savings are observed separately. At the demand side, household transportation and other household demands are also separately identified. Key elasticities of substitution between inputs used as reference in EPPA6 are depicted in Table 5 whereas Appendix A reports a detailed description of the sets and parameters of the model.

Production sectors transform primary factors and intermediate inputs into goods and services by choosing the output level that maximises profits (and minimises costs), given the available technology and market prices. Producers receive payment in return from supplying those products to domestic or foreign agents, so that the balance is maintained within the economy and across trading regions[58]. International trade is accommodated via Armington assumption (Armington, 1969), with the exception of crude oil, being a homogeneous good. With the Armington formulation, goods are treated as imperfect substitutes thereby being differentiated by region of origin and having separate prices.

For production sectors, technical substitution possibilities and requirements are described in a nested fashion[59], exemplified here with the production structure of fossil fuel-based electricity generation and the energy intensive industries, respectively in Figure 6 and Figure 7. These are the two sectors covered by the sectoral ETS being modelled here. The production structures of other sectors are presented in Appendix A, as well as the specification of sets, production activities and related price index and

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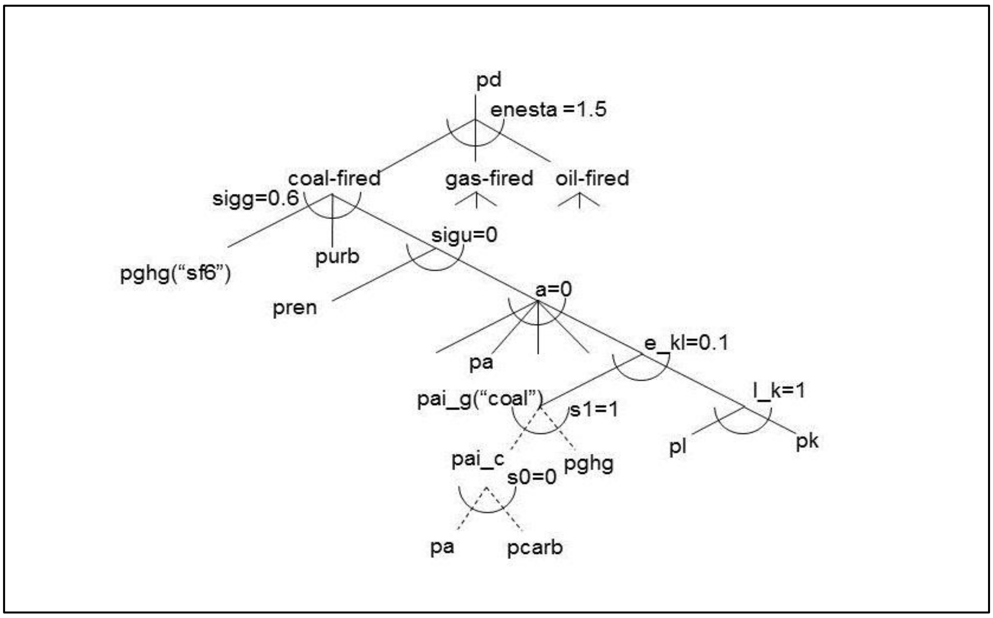
[58] It is worth mentioning that EPPA does not reflect economic rigidities that could lead to unemployment or misallocation of resources.

[59] Factor substitutability in response of changes in relative prices is only possible for malleable production. In case of vintage production, where non-malleable capital is used, the production structure becomes a Leontief (or fixed proportion production functions), implying zero elasticity of substitution.

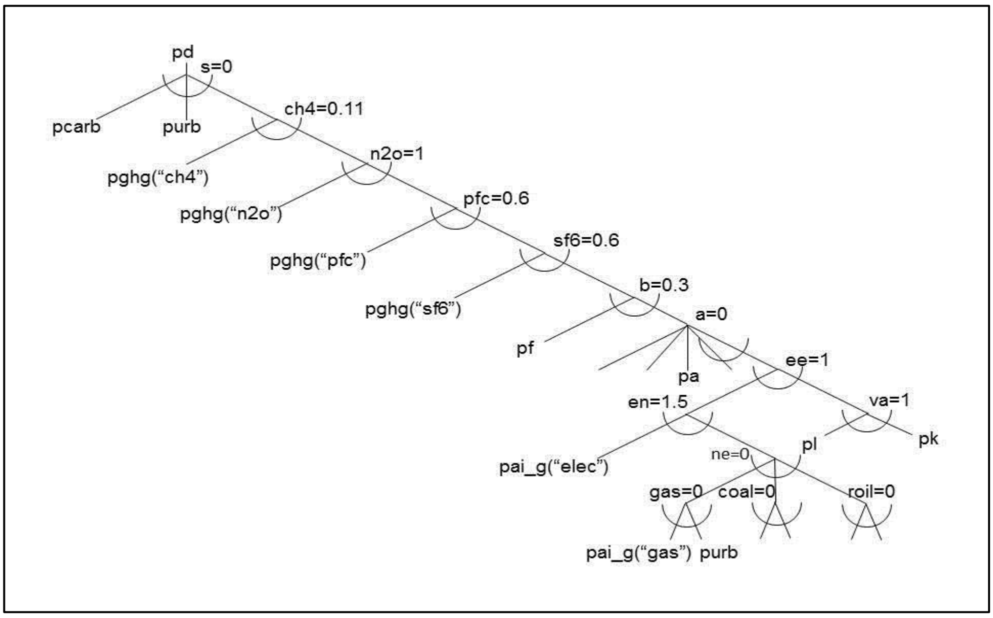
technological coefficients. Regions, sectors and backstop technologies are presented in Table 6 and Table 8.

Figure 6 – Nested structure of fossil fuel-based electricity generation



Source: Chen *et al.* (2015).

Figure 7 – Nested structure of energy-intensive sectors



Source: Chen *et al.* (2015).

Table 5 – Reference values of substitution elasticities in EPPA6

Type of substitution elasticity	Notation	Value
Between domestic and imported goods	sdm	1.0-3.0
Between imported goods	smm	0.5-5.0
Between energy and non-energy (labour-capital bundle) inputs	e_kl	0.6-1.0
Between labour and capital	l_k	1.0
Between electricity and fossil energy bundle for the aggregated energy	noe_el	0.5
Between fossil energy inputs for the fossil energy bundle	esube	1.0
Between conventional fossil fuel generation	enesta	1.5
Between natural resources and other inputs	esup	0.3-0.5

Source: Cossa (2004).

The aforementioned activities and interaction among economic agents integrate the static part of the model. As a typical CGE model formulated as a Mixed Complementarity Problem (MCP) (Mathiesen, 1985; Rutherford, 1995), all markets reach a simultaneous equilibrium in EPPA6 when the zero-profit, market-clearing and income balance conditions are satisfied. In MCP, these inequalities are related to non-negative variables, namely prices, quantities and income levels. In MCP mathematical formulation:

I. The zero profit condition can be related to cost-benefit analysis. It establishes that for any economic activity there is zero profit, which means that the value of inputs must be equal or greater than the value of outputs in all sectors of the economy. In other words, the marginal cost (MC) must equal marginal benefits (MB) when the output level (Q) is in equilibrium, as expressed below[60]:

$$MC - MB \geq 0; Q \geq 0; (MC - MB) \cdot Q = 0 \quad (1)$$

[60] Investments, imports and exports and commodity aggregation based on Armington conditions have different zero-profit conditions.

II. The market clearance condition is associated to the price level (P). As it is determined by market demand (D) and supply (S), for any good with positive equilibrium price there must be a balance between supply (S) and demand (D) and if there is excess of any good, the price must be zero:

$$S - D \geq 0; P \geq 0; (S - D) \cdot P = 0 \quad (2)$$

III. The income balance condition is formulated for income levels of households and the government and requires that the value of expenditures (E) must be equal to the value of income (I) – factor endowments and tax revenue, as follows:

$$\mathrm{E} - I \geq 0; E \geq 0; (E - I) \cdot E = 0 \quad (3)$$

This is a representation of the static component of the model, which is limited to a single period. There are some critical features to simulate the economy-environmental interactions forward in time. Such dynamics are driven by both exogenous and endogenous factors. The exogenous elements are as GDP projections for BAU growth (calibrated with Hick's neutral productivity), population growth, labor endowment growth, factor-augmented productivity growth, autonomous energy efficiency improvement (AEEI) and natural resource assets. It is assumed an increase in endowments proportional to population growth and subject to productivity growth adjustments while for the factor-augmented productivity levels, there has been a Hick-neutral adjustment to match the assumed BAU GDP growth profile of each region.

The endogenous dynamics is associated to capital accumulation (savings and investment) and fossil fuel resource depletion. Whilst savings provide funds for investing, investments of current and previous periods compose the capital for production in following periods. In EPPA6, short-term and long-term substitution possibilities between capital and other inputs are represented by vintage dynamics for malleable and non-malleable capital, as detailed in Chen *et al.* (2015). In the long-term dynamics for fossil fuels resources, depletion is set on the basis of annual production levels at each period.

Technical change is an important element of economic growth because it tends to lower the cost of production. From the energy perspective, technological developments can make advanced technologies available in order to compete with traditional energy sources due to changes in relative prices or policies that favor them. In the model, there are traditional fossil fuel-based technologies, i.e. coal, gas, oil and refined oil. In addition, alternative or new low-carbon technological options are considered as substitutes for energy commodities. EPPA6 considers 14 backstop technologies for generating low carbon energy, as listed in Table 6.

Table 6 – Backstop technologies in EPPA6

Backstop technologies	Notation
First generation biofuels	bio-fg
Second generation biofuels	bio-oil
Oil shale	synf-oil
Synthetic gas from coal	synf-gas
Hydrogen	h2
Advanced nuclear	adv-nucl
NGCC (natural gas combined cycle)	ngcc
NGCC w/ CCS (carbon capture and sequestration)	ngcap
IGCC(integrated coal gasification combined cycle) w/ CCS	igcap
Wind	wind
Bio-electricity	bioelec
Wind power combined with bio-electricity	windbio
Wind power combined with gas-fired power	windgas
Solar generation	solar

Source: Based on Chen *et al.* (2015).

In EPPA6, penetration of backstop technologies is considered to be a result of an increased demand for the output of the backstop technology over time, which is translated into more investments and growth in the supply of technology-specific factor. Eventually, the backstop technology becomes a nonbinding input as the sector expands and costs fall. In the case of advanced technologies, costs are usually higher compared to existing technologies. The economics of those technologies are described in Table 7 based on a multiplicative mark-up factor in the base year. Mark-up values are the ratio of the levelized cost[61] for each advanced technology and the cost of conventional sources in the case of electricity, while for fuels they describe the costs from that technology relative to those from existing technology wherewith they compete. For example, production costs for wind are 20% more expensive than the current technology. Assumptions for the mark-up calculations are detailed in Chen *et al.* (2015).

Table 7 - Mark-up factors for advanced technologies

Technology	Mark-up
Pulverized Coal	1.08
NGCC	1.06
NGCC with CCS	1.44
IGCC with CCS	1.55
Advanced Nuclear	1.33
Wind	1.20
Biomass	1.44
Solar Thermal	2.67
Solar PV	3.89
Wind Plus Biomass Backup	2.85
Wind Plus NGCC Backup	1.62

Source: Based on Chen *et al.* (2015).

[61] Levelized electricity cost is a measure of competitiveness that determines the price of electricity at which a certain electricity generation technology breaks even (Gavard, 2013).

This version of EPPA is calibrated using the Global Trade Analysis Project Version 8 (GTAP 8) database, with a benchmark year of 2007 (Narayanan, Hertel and Walmsley, 2012). The GTAP dataset comprises a detailed representation of national and regional input-output structure, which includes bilateral trade flows in goods and services, intermediate inputs among sectors and taxes or subsidies imposed by governments (Dimaranan and Mcdougall, 2006; Aguiar, Narayanan and McDougall, 2016). GTAP database is classified into 129 regions, 57 sectors and 5 primary factors.

For efficiency and feasibility considerations, a global model usually adopts a more aggregated level. In EPPA6, data is aggregated into 18 regions, 14 sectors and 4 factors of production, where developed and developing regions are explicitly represented, as depicted in Table 8. EPPA6 also incorporates additional data sources on energy use (IEA, 2012), energy consumption (IEA, 2012), CO_2 emissions related to cement production (Boden, Marland and Andres, 2010) and CO_2 emissions related to land use change (Riahi, Grübler and Nakicenovic, 2007). For other non-CO_2 GHG emissions and non-GHG emissions (urban pollutants)[62], data is drawn from the Emissions Database for Global Atmospheric Research (EDGAR) Version 4.2 (European Commission, 2011). However, the model has not been recalibrated to reproduce the historical projection (hindcast), and this is an area that will be considered for further research.

[62] The other non-CO_2 GHG emissions considered are CH4, PFC, SF6, and HFC and non-GHG emissions are carbon monoxide (CO), volatile organic compound (VOC), nitric oxide and NOx, sulfur dioxide (SO2), black carbon (BC), organic carbon (OC), and ammonia (NH3).

Table 8 - Aggregation of regions and sectors in EPPA6

Countries or Regions	Sectors	Factors of production
United States (USA)	*Agriculture*	Labour
Canada (CAN)	Crops (CROP)	Capital
Mexico (MEX)	Livestock (LIVE)	Natural Resources
Japan (JPN)	Forestry (FORS)	Land
Australia and New Zealand (ANZ)	*Non-Agriculture*	
Europe (EUR) [63]	Food production (FOOD)	
Eastern Europe (ROE)	Services (SERV)	
Russia (RUS)	Energy-intensive (EINT)	
East Asia (ASI)	Other industry (OTHR)	
South Korea (KOR)	Transport (TRAN)	
Indonesia (IDZ)	Ownership of Dwellings (DWE)	
China (CHN)		
India (IND)		
Brazil (BRA)	*Energy supply*	
Africa (AFR)	Coal (COAL)	
Middle East (MES)	Crude oil (OIL)	
Latin America (LAM)	Refined oil (ROIL)	
Rest of Asia (REA)	Gas (GAS)	
	Electricity (ELEC)	

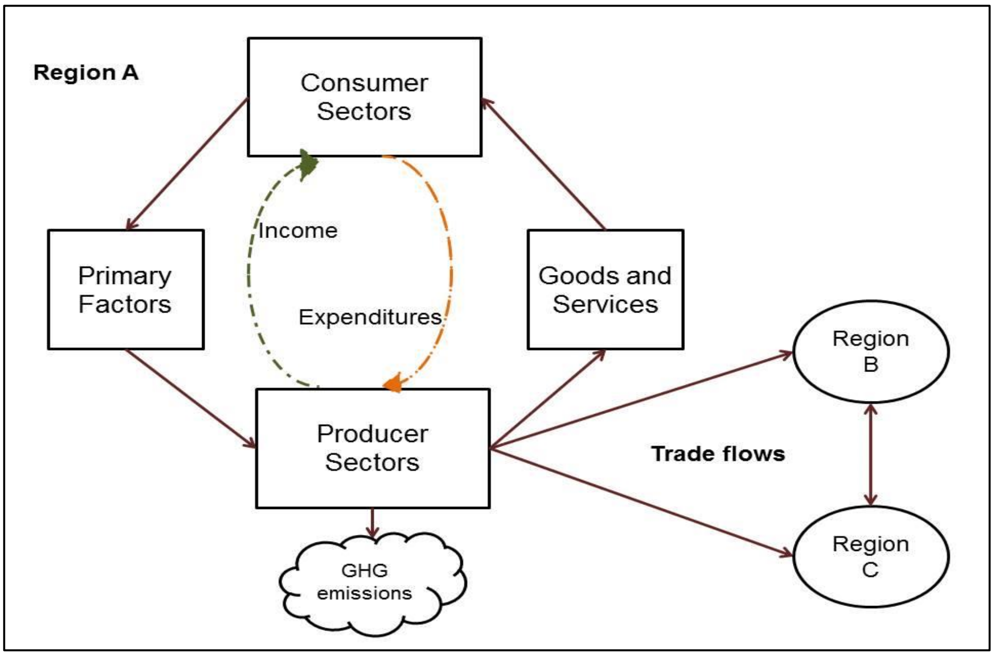
Source: Based on Chen *et al.* (2015).

EPPA6 has a very detailed characterisation of the economy, as well as the energy sector, where all GHG emissions are considered. Hence, it enables application of different mitigation and energy policies and quantification of the related effects forward in time. More specifically, it is able to incorporate emissions constraints, carbon and energy taxes or technology regulations on

[63] Europe is a region composed by the European Union (EU-27) plus Croatia, Norway, Switzerland, Iceland and Liechtenstein.

regions, gases or sectors within different policy arrangements. When an emission trading system is modelled, the flow of permits becomes part of the trade flow in goods and services thereby entering in the trade balance. The representation of the circular flow of goods and resources at both domestic and international level in EPPA is illustrated in Figure 8.

Figure 8 - The circular flow of goods and resources in EPPA



Source: Adapted from Paltsev *et al.* (2005).

A typical manner of representing an ETS is to introduce an additional constraint to limit the carbon emitted from fossil fuels to a specified rate, which creates a complementary relation between the use of fossil fuels and the correlated amount of permits. The quantity of those emission permits is sector-specific and enters in the nesting structure in combination with every type of fossil fuel energy in Leontief form. If mitigation is possible as a result of the implementation of low carbon technologies, elasticity of substitution different from zero is considered (Gurgel, 2012).

As a result of limiting emissions, a shadow value of the applied constraint is calculated. This is similar to the shadow value associated to the fixed endowments of capital, labour and natural resources. The shadow price of carbon is then interpreted as a price obtained under the potential permit market in the ETS, which is the identical representation of a carbon tax in the model. The price corresponds to a physical unit of the GHG emissions released and exhibits an increasing marginal cost, which means that, to the extent the constraint is tighter, the shadow price will increase. As a response, there is a relative change in the economics of technologies, where advanced technologies become available cost-effectively and compete with traditional energy technologies on an economic basis. ETS simulations with EPPA6 have a solution in which the least-cost abatement is achieved for each sector and type of emission, and prices are equilibrated if trading of emissions is allowed.

In EPPA6, it is possible to simulate different ETS designs in terms of sectoral, emissions or regions coverage, as well as other features such as banking of allowances or revenue recycling. To allow the banking of allowances to be included in the ETS design, the carbon price trajectory is controlled in order to reproduce a price that increases at a constant real interest rate. This is in accordance with the Hotelling model for the economics of exhaustible resources (Hotelling, 1931). The EPPA6 model considers the interest rate of long term equilibrium to be 4% per year.

As for the revenue recycling mechanism, it enters the model to reallocate the economic value of the emissions permits into other purposes rather than to the representative agent. If no specification is made, EPPA6 assumes that revenues collected from the permit trading are allocated as an endowment to the representative agent, who sells permits to sectors and consumers. It may be thought as an auction mechanism, where revenue accrues as a lump sum transfer to families. A simple way to represent revenue recycling in EPPA6 is to introduce a reduction of labour or capital taxes, which is the equivalent of providing a subsidy. Depending on how the revenue is recycled and to which sectors it is granted, it may induce further mitigation by economically stimulating a wider adoption of low-carbon technologies or alternatively, help supporting economic sectors that are adversely affected as well as alleviating regressive effects of the increase in energy costs on the representative agent.

Modelling the proposed sectoral ETS requires adjusting the model to allow sector-specific permits trading at international level. There are different ways to represent permit trading under the model functionality: i) to limit emissions trading to the domestic level, which results in sector-specific permit prices in the region, ii) to make permits tradable across sectors within regions but not across regions, which results in region-specific permit prices, or iii) to allow the trade of permits across sectors and regions, generating an international permit price. In this research, the sectoral ETS regulates carbon emissions of two production sectors in a national and international perspective. Further details on EPPA6 may be found in Chen *et al.* (2015).

4.3 Sectoral ETS and mitigation objectives

In earlier UNFCCC sessions, the main involvement developing countries had with carbon markets was through the CDM, being project hosts without binding pledges. Conversely, in the Paris Agreement both developed and developing regions affirmed long-term mitigation goals. However, a top-down global agreement has not been achieved yet. In this context, sectoral trading has been proposed as an intermediary alternative that could lead to a future global ETS approach.

For modelling the carbon policy in this thesis, we apply in Brazil a cut of 37% and 43% relative to BAU emissions by 2025 and 2030. Although the Brazilian NDC considers commitments relative to 2005 levels, for modelling purposes we define the target relative to BAU emissions (2007). Thereafter, the mitigation target is projected to rise 2% per 5-year period up to 2050, when a 50% reduction is achieved. This long-term mitigation of 50% is consistent with the proposed abatement level for Brazil in Octaviano, Paltsev and Gurgel (2014).

Although there is no explicit reference to any intention of setting up a market-based policy, irrespective of whether a cap-and-trade system or a carbon tax, the PNMC does allow the use of these instruments. In the absence of an ETS in Brazil, this study proposes an ETS design for Brazil that could facilitate linking with other schemes. The ETS design was defined to mimic the EU ETS, serving as a realistic prototype for other planned systems. Evidence suggests that the smaller system tends to cede proprietary features, which could be the case for Brazil. In addition, such upfront harmonisation is important to facilitate the linkage negotiation.

The restrictions on emissions represent the regulation stringency. The same sectoral and emissions coverage as the EU ETS are applied to Brazil so that both systems regulate electricity generation (ELEC) and energy intensive industry sectors (EINT), and only CO_2 emissions are subject to the absolute cap. The ETS sectors are assumed to be allocating tradable allowances between them. There is no specified limit on the amount of sectoral permits that can be traded.

It is worth noting that carbon emissions from land use change and deforestation are not taken into account in the simulations. From the linkage perspective, negotiation on the use of offsets is rather problematic. According to Ferreira Filho and Horridge (2017), in order to meet the Brazilian Paris' commitments, additional efforts to reduce energy emissions are required in addition to the major compromise to eliminate illegal deforestation. Currently, there are provisions to address total deforestation involving economic opportunities based on the use of carbon certificates, as established in the Brazilian Forest Code. Although it includes compensatory mechanisms for afforestation and land recovery, it is still at an early stage. Also, the inclusion of REDD+ activities in the climate governance are to be negotiated in COP-23.

Hence, given that emissions from those sectors represent a relatively high share of total Brazilian emissions, controlling emissions from those sectors in the simulations would automatically prevent other sectors from broadening mitigation effort to comply with national climate targets. In fact, the ETS coverage in most of the active systems has comprised a reduced number of sectors with large emitters of CO_2, typically the power and energy-intensive sectors, where MRV are more accurate and easier to implement. Conversely, the New Zealand ETS is a pioneer in regulating forestry and agricultural emissions, providing substantial lessons that could be used for the Brazilian case. This will be considered for future modelling work.

For the European system we applied the progressive emission reduction linear factor of 1.74% per annum from 2013-2020 and 2.2% from 2021-2030 as already specified for the EU ETS[64]. From 2030 onwards the mitigation target is assumed to increase by 1% per year until it reaches a target representing a 73% reduction of 2005 levels by 2050. In the modelling exercise no distinction is made on the EU ETS phases, the bank of unused oversupply of carbon

[64] The limitation of using the EU ETS targets is that we could not incorporate the EU commitment to reduce emissions of 40% of 1990 levels by 2030 in the model. Instead, the EU achieves approximately 38% of 2005 levels.

allowances or the existence of the New Entrants Reserve (NER 300 programme). In addition, the aviation sector is not regulated in the EU ETS[65].

For Latin America, the reduction target used in this simulation is between 5% and 15% of BAU emissions from 2020 to 2030, whilst it ranges from 30% to 40% of BAU emissions in Mexico. For modelling purposes, the mitigation target is projected to rise up to 35% and 50% emission reduction in 2050, respectively. According to previous studies (Clarke *et al.*, 2016; van der Zwaan, Calvin and Clarke, 2016), emissions reductions ranging from 20% and 50% in Latin America can provide a reasonable abatement level that is consistent with the 66% chance of keeping temperature change below 2°C. Although emissions from land use change and deforestation predominate in several Latin American countries, thereby being a critical element for abatement (Clarke *et al.*, 2016), those are not taken into account in the exercise.

In order to comply with the main objectives of the thesis, Latin America is unified as one potential trading partner, i.e. it is treated as the sum of Latin America and Mexico, as Latin America (LAM), Mexico (MEX) and Brazil (BRA) are treated as separate entities in EPPA6. As highlighted in Chapter 3, climate policy through an ETS is increasingly taking shape in the region. Examples include the recent approval in Mexico of a mandatory ETS schedule to initiate in 2020, or the explicit intention of Chile and Colombia to set up an ETS in the medium term. In the long-term, linking is also envisaged in the context of the Pacific Alliance, which includes the participation of Peru. This is in line with the official pledge made under the Paris Declaration on Carbon Pricing in the Americas[66] in 2017, which recognises the development of a carbon market at different levels as a useful and effective instrument to mitigate emissions.

Whilst there is no evidence that a comprehensive Latin American ETS will be negotiated in the short run, the Chinese ETS is already functioning. In the modelling, the mitigation target applied for China is between 45% to 60% of the GDP intensity in the 2020-2030 period, thereafter the mitigation target is

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[65] Although it is subject to the carbon tax applied to non-ETS sectors.
[66] More details in this regard are available at IETA’s website: http://www.ieta.org/resources/News/Press_Releases/2017/Declaration%20on%20Carbon%20Pricing_FINAL.pdf.

assumed to increase 5% per period so that 80% reduction of the GDP intensity is achieved by 2050[67], translated into absolute reductions. To do that, emissions and GDP were firstly obtained from the BAU simulation and then an emissions/GDP ratio was calculated. The absolute cap results from applying this ratio from the reference scenario into the emissions target.

Additionally, a supplementary policy is included by means of hypothetical (endogenous) carbon taxes on the remaining non-ETS sectors. It has been imposed to mimic other domestic abatement measures and to minimise carbon leakage[68] from ETS to non-ETS sectors. The taxes prevent carbon emissions in those sectors from exceeding BAU levels and reflect the aggregate marginal abatement costs of these sectors. The sectoral tax is generated by the model in order to induce each sector to cut emissions by the same national percentage target. Imposing a sectoral carbon tax on non ETS sectors may not be realistic, but an ETS alone is unlikely to allow a country to achieve its Paris emission reduction targets. The sectoral carbon tax captures in a simplified way the several alternative sectoral measures a country may use to mitigate emissions, given the current limitations in bringing all sectors into a single national ETS system.

All other regions in the model follow the same hybrid market approach domestically, with the CO_2 constraints being in line with their pledges under the Paris Agreement from 2020-2030, based on the information available on the UNFCCC website. Since EPPA6 is aggregated into regions and pledges are determined at a national level, the mitigation goals were defined taking into consideration the most representative country in the region where data is available, or the average of the pledges committed. From 2030 onwards, targets were estimated following the same average mitigation effort as officially committed for the Paris Agreement period. The targets applied to each region in EPPA6 are listed in Table 9 below. The majority of regions committed to a level

[67] This is consistent with projected emission reductions for China from 2020-2050 in Qi *et al.*(2015).
[68] Carbon constraints are placed on every region and the cost of producing energy intensive goods increases for all. Some leakage could still occur due to the effects of other features in EPPA (Reilly *et al.*, 2007).

of mitigation based on BAU emissions, apart from China and India which based their targets on GDP intensity. The relative stringency of these targets is therefore largely associated to GDP levels in each period.

Table 9 - Emission reduction targets[69] relative to BAU for the regions in the modelling exercise

Region	2020	2025	2030	2035	2040	2045	2050
AFR	34%	38%	42%	44%	46%	48%	50%
ANZ	24%	26%	28%	30%	32%	34%	36%
ASI	15%	17%	20%	23%	26%	29%	31%
BRA	18%	37%	43%	45%	47%	49%	50%
CAN	20%	25%	30%	35%	40%	45%	50%
CHI	45%	50%	60%	65%	70%	75%	80%
EUR[70]	14%	25%	36%	47%	58%	69%	80%
IDZ	26%	23%	29%	32%	35%	38%	41%
IND	25%	30%	35%	40%	45%	50%	55%
JPN	20%	23%	26%	29%	31%	33%	36%
KOR	30%	33%	37%	40%	43%	46%	49%
LAM	5%	10%	15%	20%	25%	30%	35%
MES	5%	8%	10%	13%	16%	19%	21%
MEX	30%	35%	40%	42%	44%	48%	50%
REA	5%	7%	10%	13%	18%	21%	23%
ROE	15%	20%	25%	30%	35%	40%	45%
RUS	25%	27.5%	30%	35%	40%	45%	50%
USA	17%	28%	38%	46%	52%	58%	64%

Source: Based on UNFCCC (2015) data for the period of 2020-2030.

[69] Most of the pledges for 2020-2030 are specified in absolute terms. However, others are subject to GDP intensity, such as China and India.
[70] This is the equivalent of 1.74% linear reduction by 2020 and 2.2% from 2021-2030.

4.4 Definition of Scenarios

We infer the economic and environmental impacts of sectoral trading in the period 2020-2050 by comparing results for the scenarios summarised in Table 10. We distinguish three perspectives: No-Policy scenario (BAU) without any mitigation policy, a domestic ETS, and a linked ETS, which is differentiated into seven scenarios (from III to VIII). The domestic ETS scenario establishes an ETS where electricity and energy-intensive industries are capped at a level consistent with the national mitigation objectives, as described in Table 9. This structure also holds for other regions of the model. Permits for polluting are obtained at national level, which generates a counterfactual carbon price.

In all international trading scenarios, allowances flow from the region with the cheapest abatement cost, thereby equalising prices and guaranteeing a cost-effective policy. Linking is differentiated in to four main aspects: the stringency of Brazilian targets, the use of flexibility provisions in the ETS design, the recycling of revenues to alleviate distributional impacts[71], and the participation of other regions through ETS commitments. More specifically: i) a linked system with no flexibility arrangements, (i.e. no banking or revenue recycling to renewables), which is the standard scenario; ii) a link considering a lower ambition for Brazil based on the same framework[72]; iii) a linked system without active market-based instruments in other regions and no flexibility rule applied to Brazil and Europe, iv) a linkage in which banking of allowances over periods is possible (it permits to shift reductions to a lower-cost time period); v) a link with revenue recycled to the production of renewable energy for Brazil and finally, vi) a link with revenue recycled as a subsidy to capital on the production of renewables in Brazil.

[71] The recycling mechanism is related to payments of permits traded in the integrated ETS system, which is assumed to be distributed through auctioning. Note that revenues from the supplementary policy on non-ETS (the carbon tax) are separated from ETS revenues, being distributed as a lump-sum to households and the government.

[72] Considering that the mitigation level implemented is relatively stringent for Brazil, with the potential to impede the linkage to take place, in this scenario we apply a reduced mitigation target which is 27% lower than the other. This is equivalent to the share of land use and deforestation emissions, which is used to infer a target focused on other economic sectors only. Rather than reducing 50% of emissions by 2050, this scenario implies a 37% reduction from BAU in the same year.

Table 10 – Summary of scenarios for the Brazil-EU linkage

No.	Scenarios	Description
I	No-policy	no mitigation policy applied
II	Bra-ETS/ EU-ETS	a sectoral Brazilian and European ETS
III	Bra-EU-Trade	a Bra-EU link, no banking, no revenue recycling to renewables
IV	Bra-EU-Ambition	a Bra-EU link with reduced mitigation ambition for Brazil, no banking, no revenue recycling to renewables
V	Bra-EU-Only	a Bra-EU link, no banking, no revenue recycling, no ETS policy in other regions
VI	Bra-EU-Banking	a Bra-EU link that allows only banking
VII	Bra-EU-Rev-RW	a Bra- EU link with revenue recycled into the production of renewables in Brazil
VIII	Bra-EU-Rev-RK	a Bra- EU link with revenue recycled as subsidy to capital on the production of renewables in Brazil

If Brazil implements a nationwide sectoral ETS, new opportunities for climate cooperation may arise, including with emerging schemes in developing countries. Since this study assumes that Brazil would decide the trading partner in advance, alternative to the developed-developing country linkage, we also evaluate a potential ETS policy proposal with China and Latin America. These linkage candidates are considered due to geographic proximity and historical economic relations, for example, via international trade of goods and services. This scope will shed light to the best policy situation, as it identifies which linkage is most beneficial and politically feasible.

Scenarios to help evaluate the best alternative in terms of partnership are described in Table 11, which follows the same structure as defined for the Brazil-EU linkage. Besides the national sectoral ETS for each partner, there are two identical scenarios that allow linking with no additional provision on the design, one considering a lower ambition for Brazil and the other with no ETS commitments on other regions, and three scenarios with banking or different possibilities of revenue recycling. Note that the revenue of permits is recycled to

Brazil in all linkages as a compensatory measure for the distributive impacts, since it has the most stringent target.

Table 11 – Summary of scenarios for the linkage with alternative partners

Scenarios			Description
I. Regional link	IX	Bra-ETS/LA-ETS	a sectoral Brazilian/Latin American ETS
	X	Bra-LA-Trade	a Bra-LA link, no banking, no revenue recycling to renewables
	XI	Bra-LA-Ambition	a Bra-LA link with reduced mitigation ambition for Brazil, no banking, no revenue recycling to renewables
	XII	Bra-LA-Only	a Bra-LA link, no banking, no revenue recycling, no ETS policy in other regions
	XIII	Bra-LA-Banking	Bra-LA link that allows only banking
	XIV	Bra-LA-Rev-RW	a Bra- LA link with revenue recycled into the production of renewables in both regions
	XV	Bra-LA-Rev-RK	a Bra- LA link with revenue recycled as subsidy to capital on the production of renewables in both regions
II. BRA-CHN link	XVI	Bra-ETS/CHN-ETS	a sectoral Brazilian/Chinese ETS
	XVII	Bra-CHN-Trade	a Bra-CHN link, no banking, no revenue recycling to renewables
	XVIII	Bra-CHN-Ambition	a Bra-CHN link with reduced mitigation ambition for Brazil, no banking, no revenue recycling to renewables
	XIX	Bra-CHN-Only	a Bra-CHN link, no banking, no revenue recycling, no ETS policy in other regions
	XX	Bra-CHN-Banking	Bra-CHN link that allows only banking
	XXI	Bra-CHN-Rev-RW	a Bra- CHN link with revenue recycled into the production of renewables in both regions
	XXII	Bra-CHN-Rev-RK	a Bra- CHN link with revenue recycled as subsidy to capital on the production of renewables in both regions

For some of these scenarios we provide a sensitivity analysis on sectoral CO_2 emissions and CO_2 prices, GDP and welfare under different assumptions for growth and autonomous energy efficiency improvements. This is important because in long-term projections, major drivers of future emissions and energy levels as well as carbon prices, are parameters subject to uncertainty such as economic growth and energy efficiency. In the sensitivity tests, there is a 20% variation from the values used in EPPA6. The “high” scenario has a representation 20% above the base level for GDP, for example, whereas “low” is interpreted as 20% lower.

The sensitivity analysis is useful to assess the assumptions made, being presented in Appendix E. Chapter 5 presents the results and examines the policy proposal using environmental (emissions and energy), economic and distributional impacts as evaluation criteria. Further, it discusses compatibility and feasibility, with the aim of understanding to what degree international coordination via an ETS may cope with several political challenges without undermining sustainable development.

CHAPTER 5
THE EFFECTS OF BRAZILIAN CARBON MITIGATION AND TRADING STRATEGIES WITHIN A LINKING FRAMEWORK UNDER DIFFERENT CONTEXTS

5.1 Introduction

This chapter presents results of the simulations with EPPA6 for the scenarios previously defined. Results reflect both the core assumptions of the model and the design of the market mechanism in which the ETS and linkage architecture take place. International cooperation is structured by linking sectoral ETS programmes at national level.

This thesis assumes that Brazil may decide whether or not to undertake a bilateral climate agreement, and which partner to link with, in advance. In the context of heterogeneous trading partners, the benefits of linking ETS systems may diverge, particularly because differences in both macroeconomic and energy generation profile among potential participants may affect the outcomes. Ultimately, development level differences are assumed to play a role in deciding on the partner for the link.

For the purpose of this analysis, such heterogeneity of the trading candidates is firstly investigated in order to identify differences compared to Brazil, serving as a basis for the discussion on linkage proposals. The second part evaluates the effects of Brazilian carbon mitigation and trading strategies within a linking framework under different contexts. The aim is to characterise the extent to which a proposed partnership may be appropriate. Further, it intends to shed light to what degree international coordination via an ETS may cope with several political challenges of transitioning to a low-carbon economy without undermining sustainable development.

5.2. Macroeconomic profile and energy pattern before policy implementation

A climate policy should be appropriate for national circumstances, and economic specificities play a major role in influencing the capacity to mitigate, as well as overall performance. The climate coordination is proposed among countries with different macroeconomic structures, as exhibited in Table 12 from baseline output in 2015. This 2015 information is taken from EPPA6, which derives information from the IEA and GTAP database.

Table 12 - Macroeconomic and emissions statistics for Brazil and potential trading partners (Europe, Latin America and China) in 2015[73]

Indicators	Brazil	Europe	Latin America	China
GDP per capita (US$)	8078.5	34579.4	7364.4[74]	4829.3
GDP (US$ billion)	1645.3	18272.1	3088.5	6806.8
Consumption (US$ billion)	968.2	11541.9	1978.0	2665.4
Investment (US$ billion)	26.6	421.2	64.0	258.1
Government Expenditures (US$ billion)	36.1	349.0	35.3	89.0
Exports (US$ billion)	19.5	195.0	69.0	194.5
Imports (US$ billion)	14.5	292.2	57.3	127.5
Carbon intensity	0.68	0.19	0.65	1.42
Total CO2 emissions – including land use (Mt)	1114.2	3545.7	2001.0	9673.5
Land use change emissions (Mt)	712.3	-214.6	655.1	168.5

* If emissions from land use change are excluded, carbon intensity is 0.24 and 0.18, 0.44 and 1.40, respectively.

There is a substantial heterogeneity between the analysed regions. The economy of Europe is stronger, being of course, a wealthier region with lower

[73] The values for macroeconomic variables are expressed in terms of US$ in the base year, which is 2007.
[74] This is sum of Latin America and Mexico's GDP over the sum of population.

carbon intensity than Brazil. Compared to Latin America[75], the economy of Brazil is very representative with similar carbon intensity, which is still lower than that observed in China. Interestingly, investments are lower than public spending in Brazil, which performs an export-led trade pattern. In contrast, only Europe presents a deficit in trade balance.

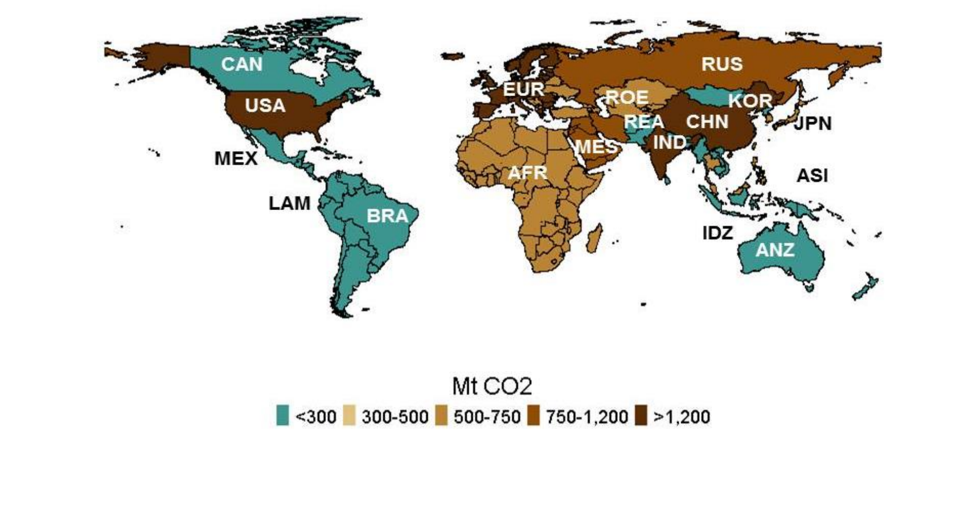
The energy use pattern is closely related to the emissions profile. A significant share of total emissions from Brazil and Latin America accrues from land use change, as depicted in Table 12. At the same time, energy use of other economic sectors contributes to overall carbon emissions. In Table 13, CO_2 emissions from coal, refined oil (roil) and gas are displayed by sector for each participant. Europe and China are the greatest emitters, even though a similar pattern of carbon emissions can be observed in Brazil and Latin America i.e., combined emissions from ROIL, ELEC, EINT and TRAN sectors correspond to 90.8%, 91.1%, 91% and 94% respectively in Brazil, Europe, Latin America and China. In this case, only in Brazil do the EINT emissions exceed those from the ELEC sector. In fact, emissions from these two ETS sectors (henceforward called sectoral emissions) in Brazil is less than 150 million tonnes of CO_2, which is the lowest level in comparison to trading candidates and other worldwide regions, as illustrated in Figure 9.

Table 13 - CO_2 emissions from fossil fuels and share (%) per sector in 2015

	Brazil		Europe		Latin America		China	
	Share	Emissions*	Share	Emissions*	Share	Emissions*	Share	Emissions*
CROP	3%	9.3	1%	25.6	2%	19.9	1%	62.6
LIVE	1%	3.6	0%	11.9	1%	5.2	1%	44.6
FORS	0%	0.8	0%	2.6	0%	1.3	0%	5.9
FOOD	1%	3.6	1%	42.9	1%	12.0	1%	68.6
ROIL	**26%**	86.3	**19%**	574.6	**15%**	153.7	**21%**	1713.7
ELEC	**7%**	23.7	**33%**	1022.2	**26%**	268.9	**53%**	4407.0
EINT	**26%**	84.0	**14%**	440.3	**19%**	195.6	**16%**	1327.6
OTHR	3%	8.9	2%	57.7	3%	30.9	2%	206.5
SERV	1%	3.9	4%	132.9	2%	24.6	1%	113.4
TRAN	**32%**	103.9	**25%**	757.5	**31%**	326.5	**4%**	357.6

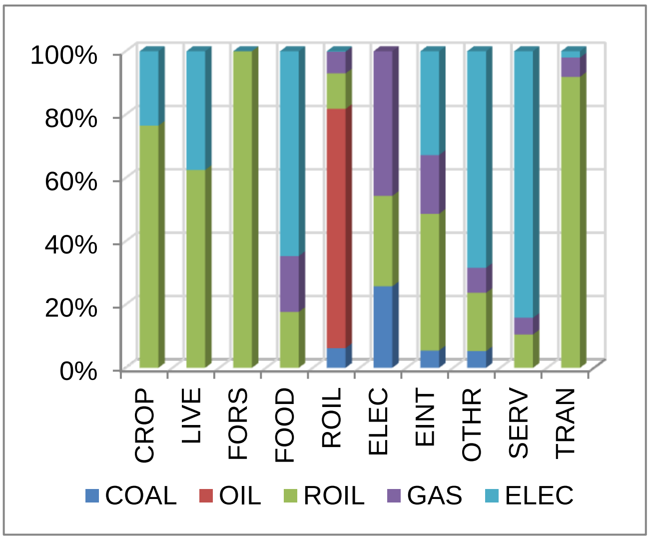
[75] Latin America hereafter refers to both LAM and MEX sectors in EPPA6.

Figure 9 - CO_2 emissions from electricity and energy-intensive sectors in 2015

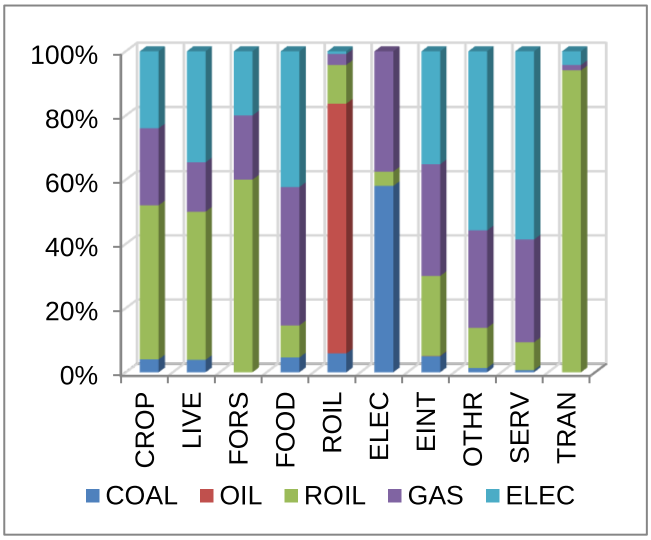


According to Figure 10(a), among the two regulated sectors in Brazil the energy-intensive sector relies more on the consumption of refined oil, electricity and gas, implying higher levels of emissions. In turn, the electricity sector is considered to be more decarbonised than other regions, particularly Europe and China as shown at 10(b) and 10(d), with approximately 2EJ of energy consumption. This is in line with evidence in Figure 11, where 49% of the total amount of primary energy used in Brazil is from low carbon technologies, particularly hydro.

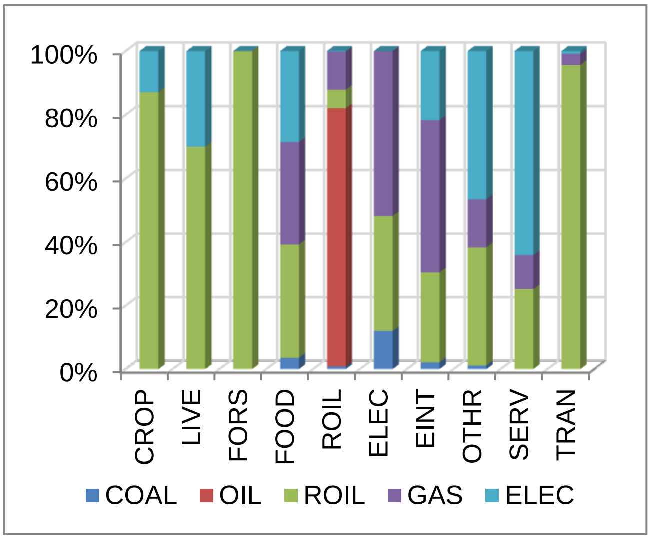
Figure 10 - Energy consumption by sector (a) Brazil, (b) Europe, Latin America (c) and China (d)



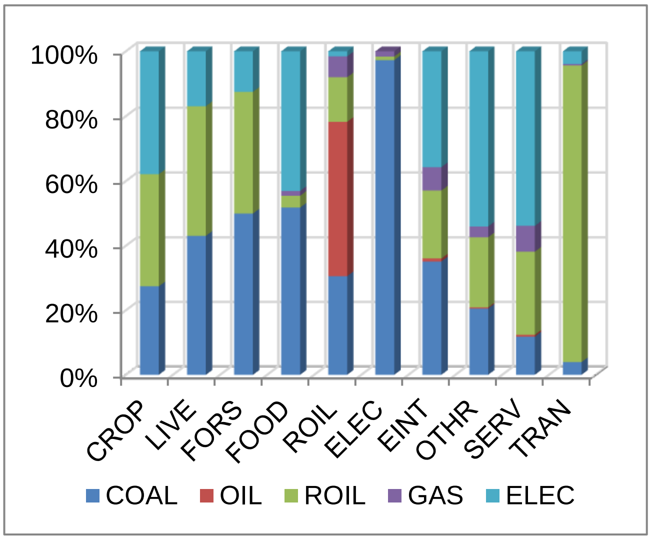
(a) Brazil



(b) Europe



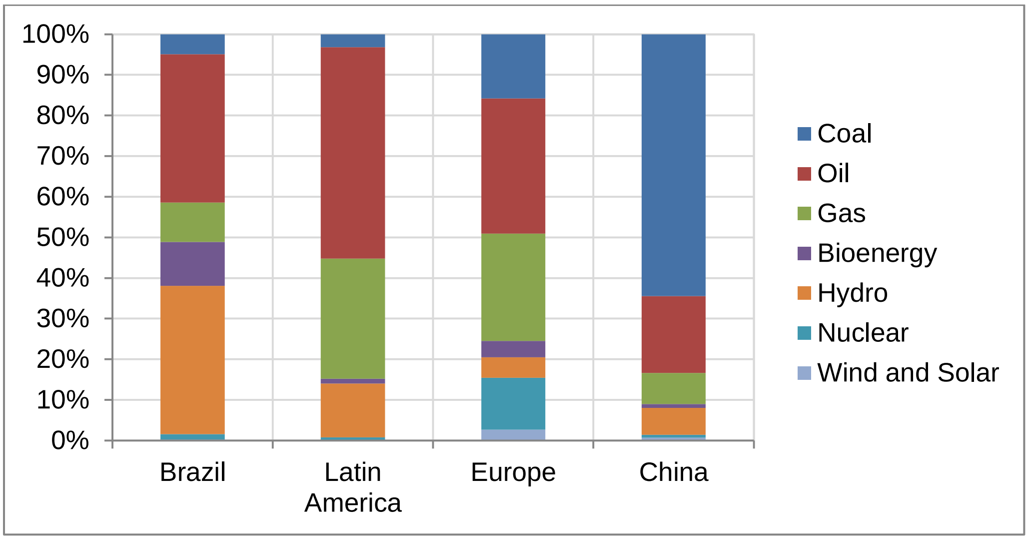
(c) Latin America



(d) China

Conversely, the majority of energy consumption of the European electricity sector comes from coal and gas, which totals 13.1 EJ. The energy mix in Europe is relatively diversified, where a substantial international trade of energy takes place to provide for economic activity. From the total use of 70.2 EJ, 75% is fossil fuel-based, with 33% of oil. Among alternative technologies, the most representative is nuclear technology, with 8.9 EJ. Of particular note is the distinct scale of energy consumption between the regulated sectors in Brazil and Europe. For example, in the case of the EINT sector in Europe, energy consumption totalises 10.9 EJ, while the sector in Brazil consumes only 1.9 EJ. As for non-ETS sectors, refined oil has the greatest importance, especially for the transport sector, which is another emissions-intensive sector in both regions and also in China and Latin America.

Figure 11 - Primary energy use in 2015



There is a similar distinction in the scale of energy consumption of the ETS sectors in China and those in Brazil. In both Chinese electricity and energy-intensive sectors, the use of coal predominates, corresponding to 47.9 EJ and 9.2 EJ, respectively. The economy of China has a very carbon-intensive energy mix comparatively, as 91% of energy used is fossil fuel-based. On the other hand, Latin America has relatively limited energy consumption, totalising 23.3EJ[76] of which only 15% in from alternative technologies. The most noticeable difference compared to Europe and China is that coal has a reduced

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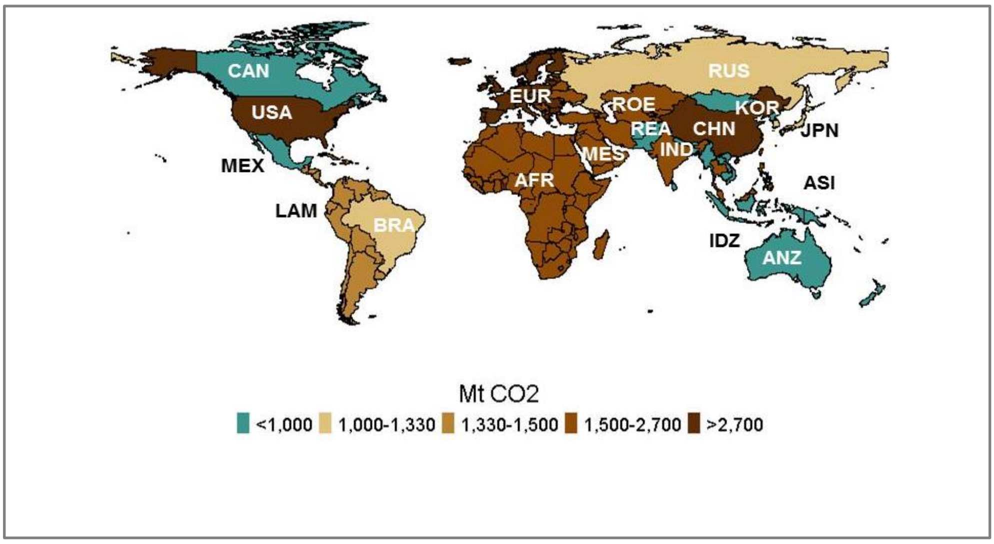
[76] The share of oil, gas and hydropower in total energy generated is significant in Latin America. According to Clarke *et al.* (2016), most countries in Latin America have a minimum reliance on coal. In terms of natural gas consumption, Argentina and Mexico present the largest share in the energy mix. On the other hand, Brazil and Colombia are more reliant on the use of hydropower.

importance in the energy mix of Latin America, which implies a limited abatement potential of rolling back its consumption.

Among the investigated countries, Brazil displays a unique energy mix, basically due to the national production and consumption of hydroelectricity and oil. This ex-ante analysis has highlighted discrepancies in the volume and sources of emissions of Brazil and those in the considered trading regions. In a worldwide perspective depicted in Figure 12, Brazil represents 3% of global CO_2 emissions whereas Europe, China and Latin America contribute to 10%, 26% and 6%, respectively.

Figure 12 - Total CO_2 emissions in 2015



Under a carbon emissions constraint, changes in energy prices tend to spur low-carbon technological developments and consequently, facilitate emission reduction obligations differently in these countries. Yet, the degree of mitigation will depend on different factors, with economic costs being the most relevant as it affects abatement from the electricity sector and energy-intensive industries. Overall implications of the sectoral ETS and linking framework are presented as follows.

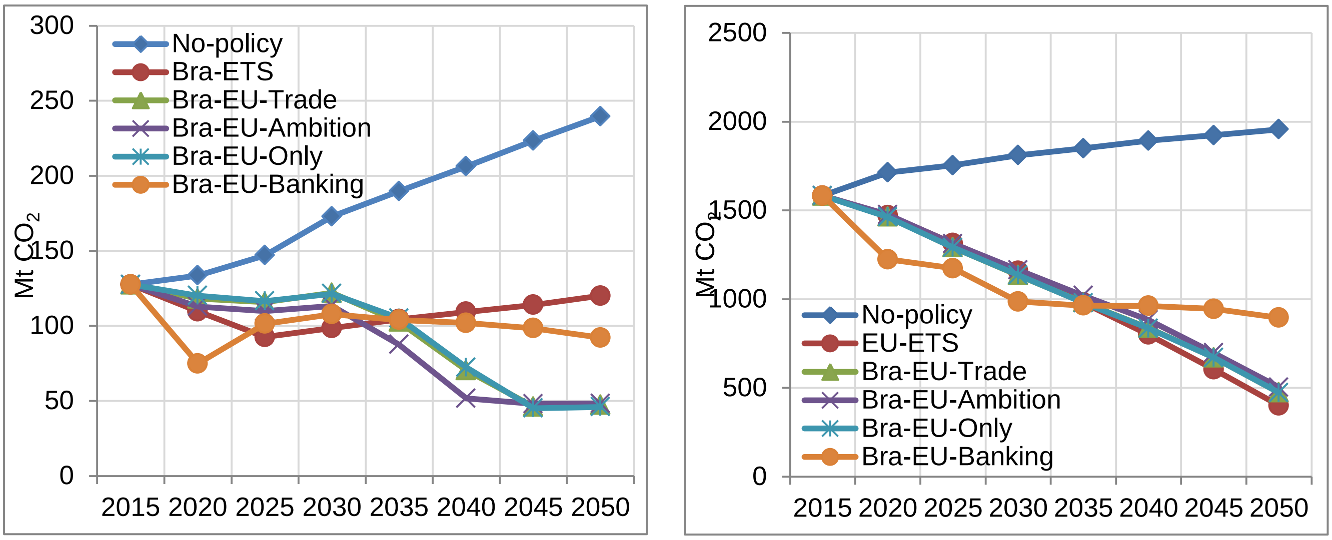
5.3 Sectoral ETS linkage under different contexts

5.3.1 A Brazil-EU ETS linkage

Emissions in Brazil and Europe from the sectoral ETS for the simulated scenarios are presented in Figure 13 and Figure 14, while carbon prices are displayed in Figure 15. Overall abatement costs of the climate policy for regulated sectors are affected by the carbon price, whether in autarky or in a linked-ETS situation. The difference is that sectoral trading leads to a carbon price equalisation between the jurisdictions involved, eliminating marginal abatement cost divergences.

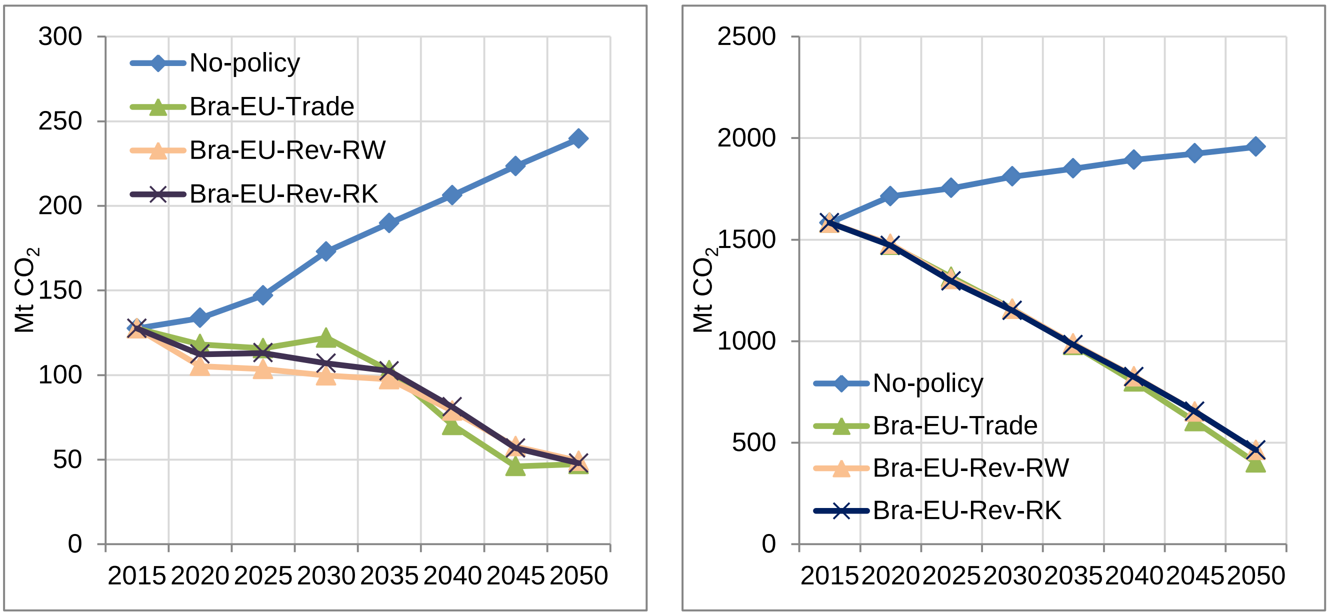
Carbon emissions from the Brazilian sectoral ETS represent less than 1% of total global emissions. In the absence of an international carbon trading system (Bra-ETS scenario), emissions from power sector and energy-intensive industries in Brazil are 98.6 and 120 million tonnes in 2030 and 2050, respectively. This is equivalent to 74.3 and 119.6 million tonnes less than NO-POLICY emissions for the same period, with a corresponding CO_2 price of US\$202.4/tCO_2 in 2030 and US\$304.9/tCO_2 in 2050. In comparison with Gurgel and Paltsev (2014) and (Domingues, Magalhães and Carvalho, 2014), this carbon price is considerably high, whereas it remains below the estimated price of US\$301/tCO_2e in (Castro, 2013) for 2030. The cap and ETS design specified along with the model used are the main drivers for these differences. A complementary analysis is provided in Appendix B focusing solely on the Brazilian ETS under alternative scenarios.

Figure 13 - ETS emissions for Brazil under different ETS design scenarios



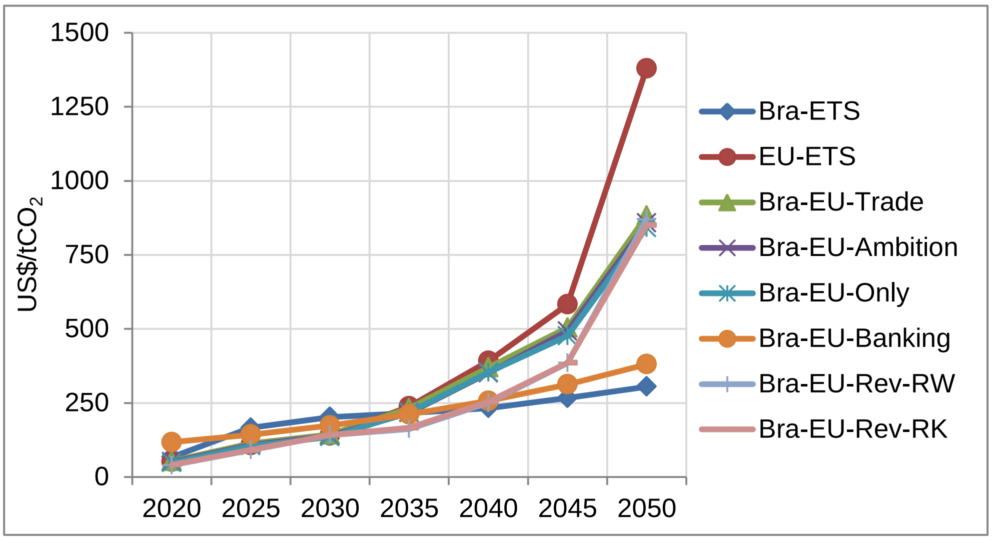
(a) Brazil (b) Europe

Figure 14 - ETS emissions for Brazil under different revenue recycling mechanisms



(a) Brazil (b) Europe

Figure 15 – CO_2 prices under different scenarios



Under the ETS constraint and without climate cooperation, sectoral emissions in the EU ETS are 1.16 and 0.4 billion tons of CO_2, or 5.8% and 2% of global emissions in 2030 and 2050. In this scenario, carbon permits cost US\$139/tCO_2 and US\$1380/tCO_2, respectively. This ETS price is endogenously derived and strongly impacted by the model representation regarding macroeconomic assumptions, availability and costs of backstop technologies, uncertainties and other modelling characterisation. Mitigation in the long-term would, indeed, require an increasing carbon price to discourage intensive reliance on carbon-based energy sources (Edenhofer *et al.*, 2009). Previous projections using different models and ETS design estimate a carbon price ranging between 120 and 1200 €/tCO2 in 2050 to meet the climate goals (Peñasco and Río, 2015; EU, 2016).

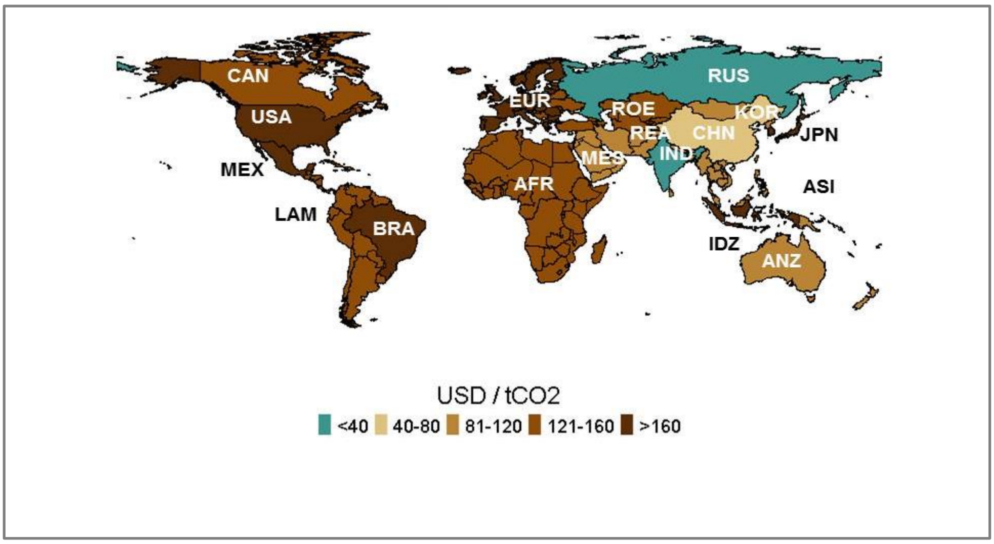
If trading is allowed between Brazil and Europe, the carbon price is equalised across the two systems at US\$143/tCO_2 and US\$142/tCO_2 in 2030 of the Bra-EU-Trade and Bra-EU-Rev-RW scenarios. These linking prices are almost pegged to the EU's autarky price of US\$139/tCO_2 in 2030, given its sheer size relative to Brazil's (in terms of volume of covered emissions), thereby making marginal abatement costs not much lower than in Europe. Also, similar prices reflect the lack of opportunities to trade allowances in this period.

Thus, Brazil faces a CO_2 price reduction when linking by replacing high-cost emissions abatement options domestically with low-cost options in Europe. The scope for replacement is enhanced by the larger volume of electricity and energy-intensive emissions in the EU ETS. Brazilian sectoral emissions in the Bra-EU-Trade and Bra-EU-Rev-RW scenarios are 122 and 99.7 million tonnes, and those from the EU ETS are 1135.3 and 1157.6 million tonnes in 2030, a reduction of 37.3% and 36.1% in BAU emissions, respectively. Note that among linked scenarios, recycling revenue in the production of renewables leads to the largest emission reductions in 2030. Since the income from permits is fully distributed to Brazil, it encourages further abatement.

Linking Bra-ETS with the EU-ETS makes a tonne of CO_2 cheaper to Brazil than obtaining it at domestic level by 2030. For instance, the minimum cost of carbon possible is achieved when only Brazil and Europe commit to mitigation via ETS (US$135/t$CO_2$) – Bra-EU-Only - or either if Brazilian ambition is lowered (US$139/t$CO_2$) – Bra-EU-Ambition. From 2035 onwards, carbon prices in a linked situation are higher for Europe relative to autarky. This is understandable as the carbon price in the EU-ETS case is greater than in all Bra-EU scenarios, reflecting its deeper effort to curb emissions from 2035 to 2050.

In a global perspective, this price of carbon permits in the Brazil-Europe linkage is among the most expensive for 2050, as illustrated by Figure 16, with exception of the Bra-EU-Banking scenario, where the use of banked allowances has gradually increased over the period, resulting in a lower cost of US$380/t$CO_2$. Results also highlight that inter-temporal permit trading appears to provide strong incentives for early action, but at the risk of surrendering additional allowances in the future. In other words, it may create an over-allocation in subsequent periods and therefore, limited reductions. In fact, it demonstrated the ability to foster carbon price stability over the period, being mostly indicated for the period 2040-2050 when resulting carbon prices are very high in other linking scenarios.

Figure 16 – A global perspective of ETS carbon prices in 2050 for Bra-EU-Trade scenario



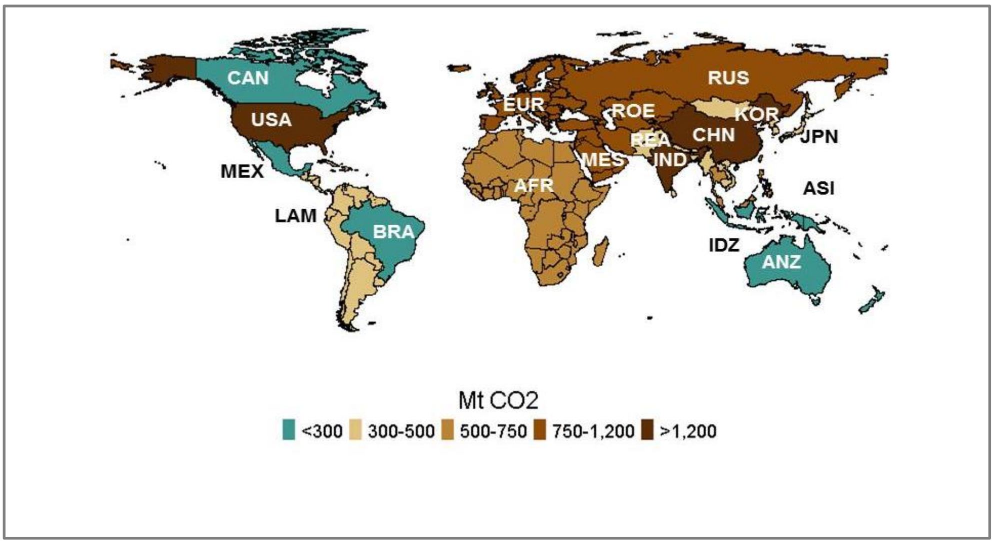
To date, carbon prices have remained persistently low in the EU ETS since it was launched, roughly hovering between €4 and €10 euros during the current third trading phase. In general, this continuous downward trend has ensued from the economic recession and renewables-promoting policies that contributed to a decrease in permit demand as well as low capacity of the system to respond to changes in economic circumstances (Grosjean, Acworth and Flachsland, 2014; Ellerman, Valero and Zaklan, 2015). This price is considered to be very low to promote significant incentive for polluters to undertake necessary investments in low-carbon technologies, to drive low-carbon innovation but also to cost-effectively achieve proposed mitigation, particularly in a context of persistent supply imbalance of carbon permits[77] (Kollenberg and Taschini, 2016).

Further, these low carbon prices are far below most estimates of the social costs of carbon (Anthoff and Tol, 2013; Foley, Rezai and Taylor, 2013) as

[77] After incurring volatile prices and windfall profits, regulators started reviewing the system so as to strengthen the functioning of the EU ETS, for example, by addressing the oversupply problem. For that, EU regulators proposed a “back loading”, that is, a reduction on the number of allowances available in the market through near-term auctions, whereas the quantity removed is later on reintroduced. Another reform incorporated was the implementation of the market stability reserve in an attempt to create a system more resilient to supply-demand imbalances.

well as not being at a meaningful level to drive deep decarbonisation. As such, sectoral emission reductions and resulting carbon prices in this simulation are coherent with the intended internalisation of the costs of pollution, although a rise in the EU's carbon price is uncertain to predict, at least in the short term.

Figure 17 – Sectoral CO_2 emissions in the Bra-EU-Trade scenario in 2030



From the climate perspective, aggregate emission reductions are a major indicator of environmental benefits. According to Figure 17, electricity and energy-intensive sectors in the EU are the largest emitters in the world along with China, India and USA. In a combined framework, Brazil and Europe account for a significant share of carbon emitted globally, i.e. 3.5% of total emissions and 6.3% of power sector and energy intensive industry global emissions in 2030. By 2050, this aggregate share reduces to 1.6% of total emissions and 2.6% of ETS emissions in a global perspective. From the simulations, the joint emission reductions over the period of compliance in the linked system are greater if potential revenues are recycled back to the production of renewables in Brazil, particularly in the long term, representing a 48% reduction compared to 2020-2050 emissions of the No-Policy scenario. The advantage of this scenario is that distributional implications from the ETS implementation in Brazil can be, to a certain extent, addressed or at least partially compensated.

Similar aggregated mitigation of the sectoral ETS is observed if the ETS linkage is designed without the use of flexible mechanisms or ETS policies in other regions, with the level of emissions in both cases being 47% lower than the No-Policy scenario. This indicates that the positive environmental effects realised in Brazil and Europe, of adopting an integrated price-based climate policy, do not depend on the existence of sectoral trading mechanisms in other regions. The banking of allowances simultaneously generates the largest aggregate abatement in the ETS in 2030 and the lowest in 2050, as expected.

At the intra-jurisdiction perspective, results indicate a substantial reduction in emissions from the electricity sector in Brazil in all scenarios, as it has more alternatives for substituting fossil fuel-based energy sources with renewable energy. It is observed that, the greatest mitigation level is achieved by the electricity sector in 2030 when in autarky, as it almost fully decarbonises in the simulation. This level of decarbonisation reflects the relatively higher ambitious target that Brazil, alone, is committed to in the context of a domestic ETS in our simulations.

Conversely, a linking agreement enables the total cap, and required abatement of participating jurisdictions to be higher than within an isolated trading system. In Brazil, there is a more profound cut in emissions of both regulated sectors in the scenarios of Bra-EU-Rev-RW and Bra-EU-Banking. This effect results from the total reduction in the use of refined oil for power generation, and coal and gas in energy-intensive industries, which face higher mitigation costs and have less carbon abatement options or technological alternatives for energy substitution. In light of progressive increases of carbon prices, the usage of banked allowances is widely adopted by energy-intensive sectors in the Bra-EU-Banking scenario. In the long-term, mitigation in Brazil is deeper in Bra-EU-Rev-RK and Bra-EU-Only scenarios, where further abatement is driven by reducing the use of refined oil in 2050.

In other words, the abatement is reallocated from Europe to Brazil in those scenarios, thereby reducing the amount of permits Brazil purchases by 2030 and increasing its sales from 2035 onwards. Given the level of stringency applied in Europe, linking promotes more emission reductions in the region than a domestic sectoral ETS, notably if allowances can be banked in the short run in light of the cheaper abatement opportunities encountered. However, the level of abatement for covered sectors in Europe remains very similar in all linking circumstances. Similar to Brazil, the majority of efforts in Europe fall on the electricity sector over the period. In the Brazil-Europe linkage, the size of the market for allowances is relatively small by 2030 due to differences in costs and mitigation opportunities between the two regions, besides the target individually implemented. In the long-term, the integrated carbon market becomes larger for the same motives.

By imposing a carbon constraint, electricity price increases and raises the costs of other sectors, which lower the output level. It is worth noting that non-ETS sectors are included in the policy framework modelled, thereby being subject to the national target. Therefore, to the extent which the sector depends on electricity as input, or the existence of other relevant emissions sources, the decline of activity levels of those sectors may also stem from the sectoral tax imposed.

Taking the domestic ETS, and the standard linking scenario (Bra-EU-Trade) to illustrate this point, Figure 18 shows the significant negative effects on the activity level of those economic sectors in Brazil. At sectoral level, there are substantial losses in output from the refined oil, oil and gas sectors. From the production chain perspective, these sectors are mostly affected by the reduced use of fossil-fuel due to the higher prices that result from the internalisation of the emissions costs.

For the ETS sectors, the increased cost of production has implications both domestic and internationally since domestic goods become more expensive relative to foreign goods, suggesting a loss in international competitiveness. The volume of exports from energy-intensive sectors to other regions decreases substantially within the policy framework modelled. Whether it is a domestic ETS or a linked ETS, exports to Latin America, USA and Europe reduce the most, as reported in Table 14.

Figure 18 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-EU-Trade linking scenario for Brazil

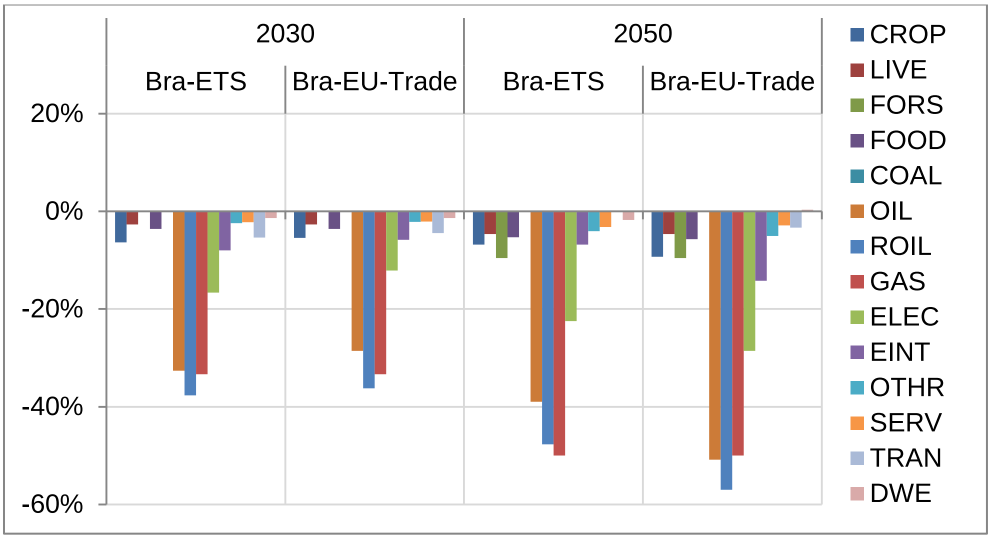


Table 14 – Changes in the volume of production exported from Brazilian energy-intensive sectors to other regions compared to No-Policy scenario

Importers	2030		2050	
	Bra-ETS	Bra-EU-Trade	Bra-ETS	Bra-EU-Trade
AFR	-5%	-3%	-6%	-15%
ANZ	-1%	0%	-1%	-2%
ASI	-4%	-3%	-5%	-14%
CAN	-3%	-2%	-4%	-10%
CHN	-5%	-3%	-6%	-16%
EUR	**-18%**	**-12%**	**-14%**	**-38%**
IDZ	0%	0%	-1%	-2%
IND	-1%	-1%	-1%	-3%
JPN	-5%	-3%	-5%	-15%
KOR	-2%	-2%	-3%	-8%
LAM	-20%	-13%	-15%	-40%
MES	-3%	-2%	-4%	-10%
MEX	-4%	-3%	-5%	-13%
REA	0%	0%	-1%	-2%
ROE	-1%	-1%	-1%	-3%
RUS	0%	0%	-1%	-1%
USA	-17%	-11%	-14%	-37%

China is also one of the major importers of energy-intensive goods from Brazil. Although there is slight reduction in 2030, the loss of competitiveness leads to a decrease in exports by 2050, especially if a Brazil-Europe linkage is active. This effect reflects the equalised cost of carbon, which also accommodates the stringent long-term environmental goals of Europe. Moreover, the impacts on exports are a consequence of the income reduction in all regions, since the internalisation of emissions increases production costs for supplying the same quantity of goods and services. Thus, in a world where every region implements climate policies, income generation as well as international trade is lower.

A carbon policy makes overall production for participating jurisdictions more expensive. Figure 19 reveals a smaller adverse effect on output of EU ETS sectors in 2030, especially the electricity sector since the cap is lower than in 2050. As a consequence of the tighter cap, the effect is deepened in the long term. Among non-ETS sectors, there are greater output losses in coal, roil and gas and, to a lower degree, in the oil and transport sectors.

Sectoral trading involves transfer of emission permits across participants. If trading is allowed between the Brazilian sectoral ETS and the European system, Brazil displays a net importer profile of carbon permits in the first decade, as displayed in Table 15, in the form of positive values. This trade pattern is derived from the relatively more aggressive reductions that Brazil is committed to in the first years of compliance, compared to Europe. Financial transfers from the Brazilian covered sectors to Europe range from US$0.2 to US$3.4 billion in the 2020-2030 period, corresponding to approximately 23.4, 14.7, 22.8, 9.2, 1.1, 8.3 and 10.1 million tonnes of CO_2 imported in 2030 according, respectively, to the Bra-EU-Trade, Bra-EU-Ambition, Bra-EU-Only, Bra-EU-Banking, Bra-EU-Rev-RW and Bra-EU-Rev-RK scenarios.

Figure 19 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-EU-Trade linking scenario for Europe

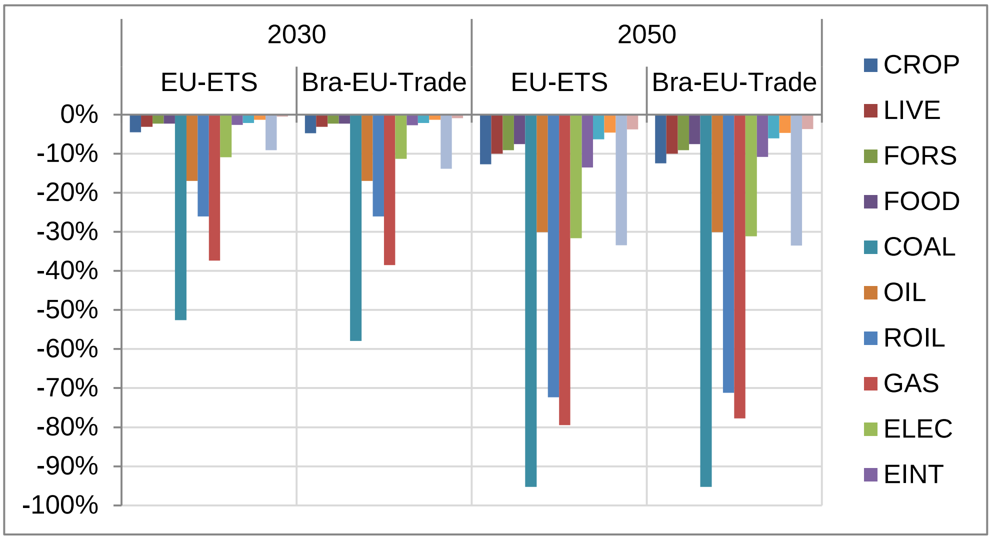


Table 15 – Total financial transfers of CO_2 permits for Brazil (in 2007 US$ billion)

Scenarios	Bra-EU-Trade	Bra-EU-Ambition	Bra-EU-Only	Bra-EU-Banking	Bra-EU-Rev-RW	Bra-EU-Rev-RK
2020	0.5	0.2	0.5	-4.1	-0.2	0.1
2025	2.6	1.9	2.5	1.2	1.0	1.9
2030	3.4	2.1	3.1	1.6	0.2	1.2
2035	-0.4	-3.6	0.2	-0.1	-1.1	-0.3
2040	-14.4	-20.2	-13.0	-1.9	-7.6	-7.2
2045	-34.2	-32.5	-32.8	-4.9	-21.6	-22.1
2050	-64.2	-61.5	-62.3	-10.6	-61.6	-61.3

A long-term linkage with a developed system such as the EU ETS implies to Brazil an emissions reduction of approximately 60% compared to autarky, totalling on average 70 million tonnes less in all linking scenarios. The only exception is the Bra-EU-Banking link, in which there is just a 23% decrease in ETS emissions relative to Bra-ETS. In this case, permits are mostly supplied by Brazil, since abatement options or technological alternatives to mitigate become more available there, besides the modest targets implemented from

2035 onwards. Hence, it receives between US$1.9 and US$14.4 billion in 2040 and US$10.6 and US$64.2 among the simulated scenarios in 2050.

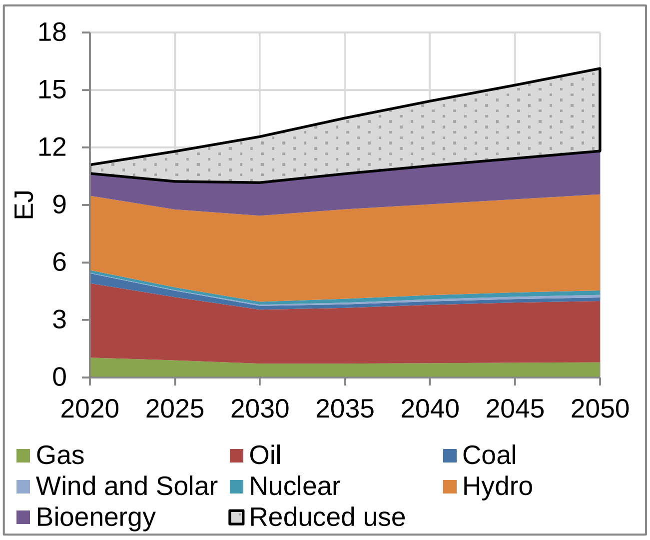
From the scenarios analysed, an inter-jurisdiction pattern can be detected, i.e. emissions reductions are transferred from Europe to Brazil by 2030 and thenceforward the inverse takes place, with international trading generating monetary flows to Brazil. This is aligned to the literature, which generally assumes long-term targets to be more stringent in Europe, portraying it as a buyer of emissions in carbon markets of either developed (Dellink *et al.*, 2014) or developing countries linkages (Gavard *et al.*, 2013; Gavard, Winchester and Paltsev, 2016; Doda and Taschini, 2017).

This trading pattern reveals some important insights about linking under the modelled circumstances. It suggests the level of ambition plays an important role towards defining buyers and sellers of permits in the carbon market, although both tend to be in a better condition than without linking. Even though the literature recommends the harmonisation of mitigation ambition, accommodating developing countries into a linked-system where the cut in emissions is deeper than that applied to the developed country fails to comply with the principle of common but differentiated responsibilities. If the developing country assumes an importer-profile, it tends to be potentially more affected by distributive effects at the local level. In our simulations, regardless of linking, the Brazilian sectors are similarly affected.

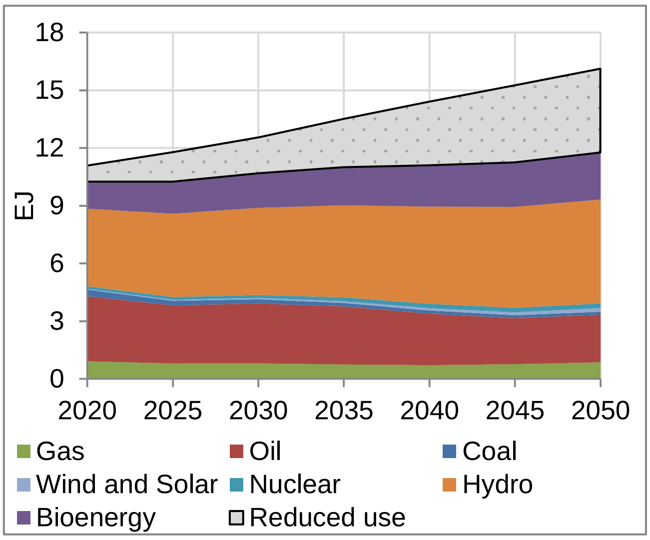
Potential revenues along with the prospects of associated cost savings can be a very attractive condition for Brazil to agree on the link. However, considering that the Brazilian electricity sector is already significantly decarbonised, with alternative energy comprising 51% of total energy used, the challenge is to move further towards increasing the share of low-carbon sources that can compete with fossil fuel-based power, especially if carbon emitters face an appropriate carbon price. In fact, Brazil is explicitly keen to strengthen its share of renewable energy in the energy mix, as stated in the NDC for the Paris Agreement.

Policies that place a price on carbon are important drivers for the adoption of environmentally friendly technologies as well as being a stimulus for low-carbon innovation (Dechezleprêtre, Martin and Bassi, 2016). Therefore, it tends to enhance substitution from a polluting economy towards a more decarbonised one by altering the demand for fossil fuels, thereby changing the energy use profile. Figure 20 shows the Brazilian energy mix in the domestic ETS situation in comparison to linking to Europe in the standard scenario (Bra-EU-Trade) and Figure 21 presents the results of primary and alternative energy use for 2030 and 2050 for Brazil in alternative ETS linkage scenarios.

Figure 20 – Primary energy use in Brazil: Domestic ETS vs. Linkage (Bra-EU-Trade)



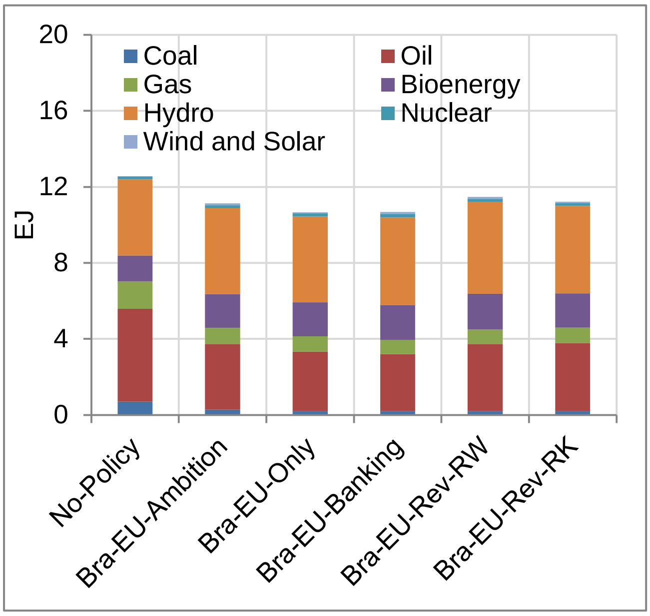
(a) Bra-ETS



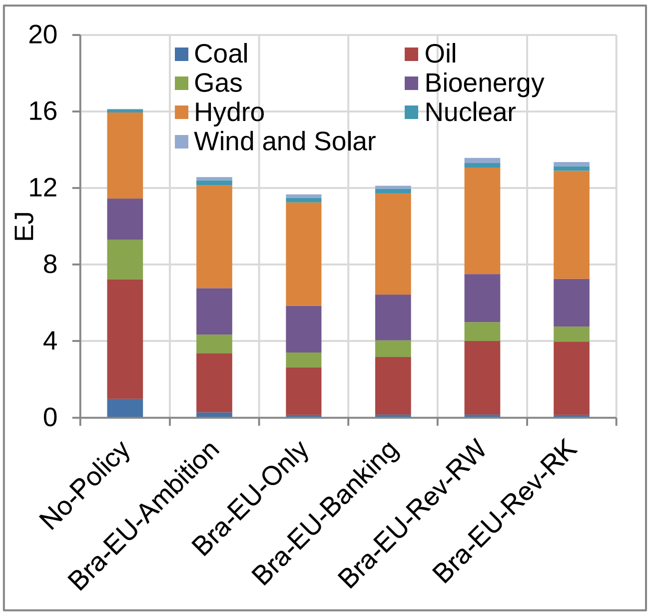
(b) Bra-EU-Trade

In the energy mix, hydro and oil predominate in Brazil, corresponding to approximately 40% and 30% of total energy, respectively, in the simulated scenarios. Total use of energy is, on average, 10.7 EJ in 2030 and 12 EJ in 2050. Overall, the total use of energy is reduced the in Bra-ETS and Bra-EU-Trade scenarios compared to No-Policy scenario. However, while a domestic sectoral ETS promotes fuel switching in the short run, linking enhances the use of low-carbon technologies for Brazil in the long-term.

Figure 21 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for Brazil



(a) 2030

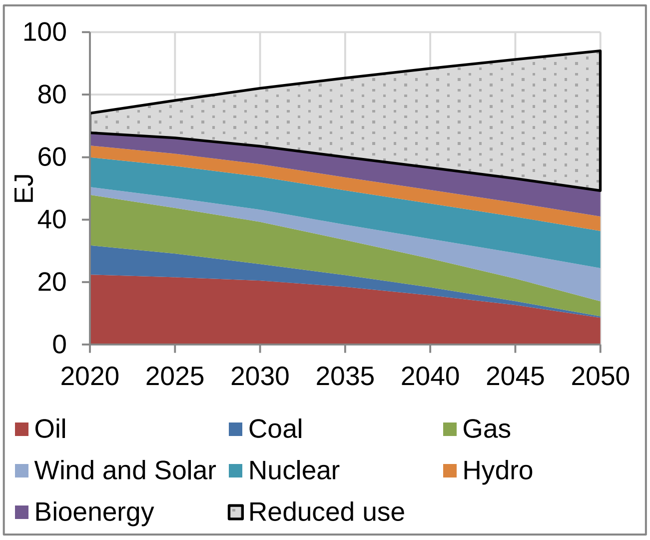


(b) 2050

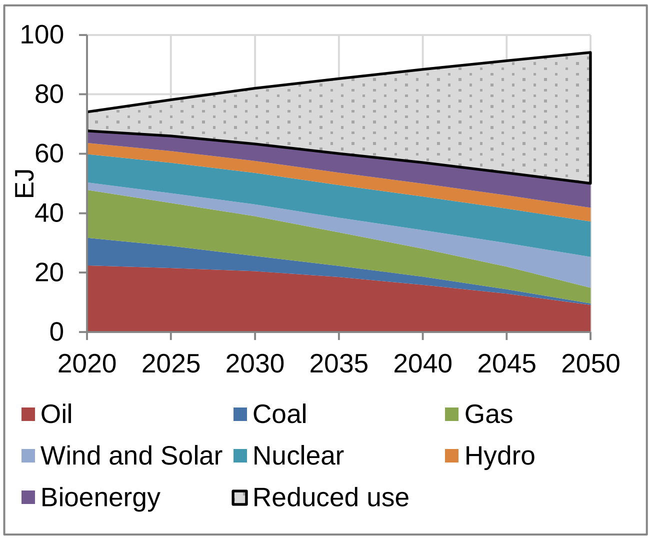
As the sectoral ETS progresses over time, the relevance of low carbon technologies becomes even more evident relative to fossil fuel-based primary energy. For instance, the share of coal decreases by 69% in the Bra-EU-Trade and Bra-EU-Only scenarios, with a further decrease of 73% when flexible arrangements are incorporated into the system (i.e. banking or revenue recycling), and with a decrease of 70% if Brazil is less ambitious in 2030. Nevertheless, in this first decade, the Bra-ETS promotes the deepest substitution towards low-carbon energy, with an alternative energy share increase to 64% of the energy mix, which is primarily due to the effort required to meet the mitigation target without any cooperation.

Among all policy scenarios simulated, whereas hydroelectricity power in Brazil faces an increment of approximately 13%, renewables (wind and solar) rise more than 6000% in relation to No-Policy scenario in 2030. Proportionally, this is still a small amount of electricity since it corresponds, on average, to only 0.09 EJ of the 6.5 EJ of alternative energy in the scenarios. The use of revenue recycling in Brazil is important particularly because it supports the decarbonisation of the energy sector the most. Similarly, the long term effect is driven by a lower demand for fossil-fuels, where primary energy use is between 3.4 EJ and 5 EJ in the policy scenarios, instead of 9.3 EJ without any mitigation target. The greatest substitution effect is verified in the Bra-EU-Rev-RW and Bra-EU-Rev-RK in 2050.

Figure 22 – Primary energy use in Europe: Domestic ETS vs. Linkage (Bra-EU-Trade)



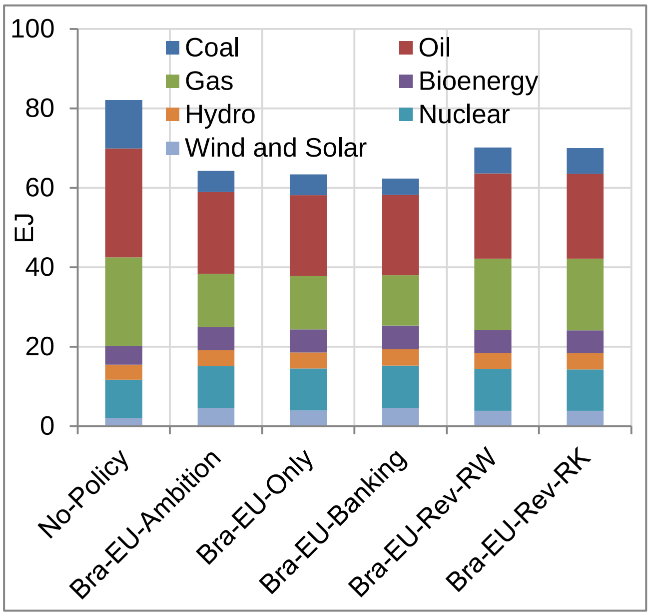
(a) EU-ETS



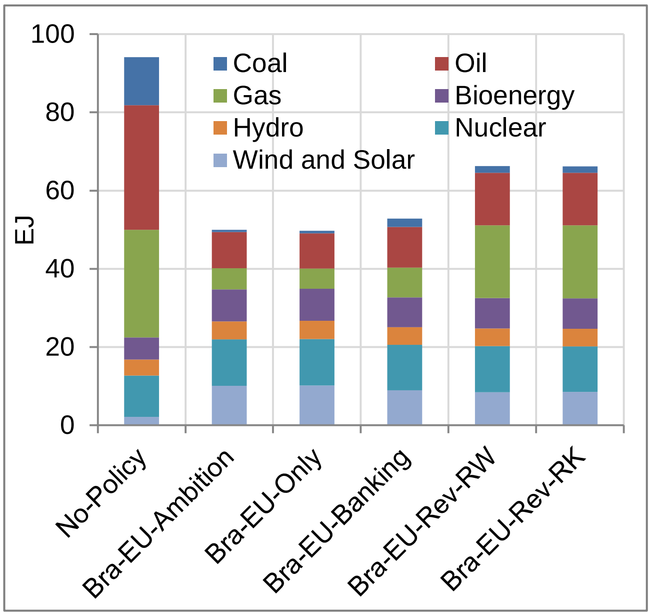
(b) Bra-EU-Trade

The European energy mix relies heavily on oil, gas, coal and nuclear energy, as depicted in Figure 22. If there is no climate policies implemented, Europe uses a total of 82 EJ in 2030, where primary energy corresponds to 61.9 EJ and low-carbon technologies account for only 25% of the total. Total energy reduction from the No-Policy scenario in the EU-ETS and Bra-EU-Trade scenarios is very similar, which save from 6 EJ to 45 EJ in the 2020-2050 periods. Technological changes are prompted by the EU ETS but linking to Brazil extends the energy substitution effect by 2030. From Figure 23, although fossil fuel energy still prevails, there is a growth in alternative energy by 21% in the Bra-EU-Trade, 20% in the Bra-EU-Only and Bra-EU-Rev-RW scenarios, 19% in Bra-EU-Rev-RK scenarios, and 23% and 25% in the Bra-EU-Ambition and Bra-EU-Banking scenarios respectively, relative to No-Policy in 2030.

Figure 23 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for Europe



(a) 2030



(b) 2050

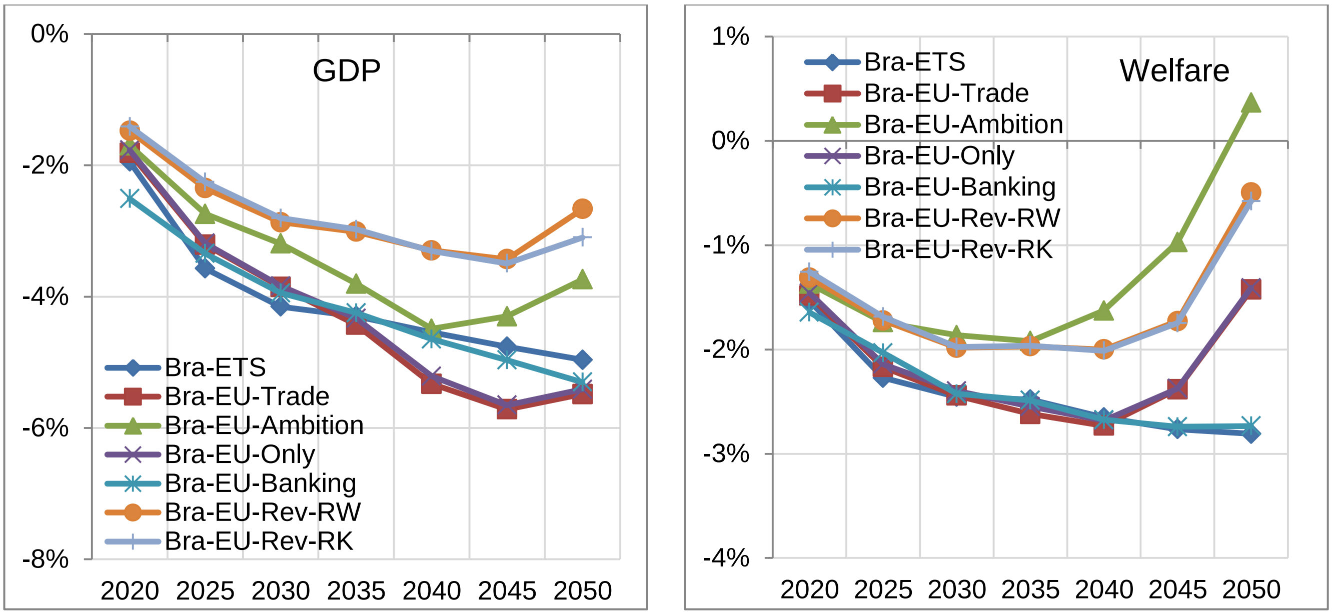
Under the proposed sectoral design, linkage enhances technological changes in Europe so that in the long term there is a substitution effect towards low-carbon sources. Among them, bioenergy and renewables use surpasses the increase in hydro and nuclear. On the other hand, the greatest reductions occur in coal and gas use. Fossil-fuel substitution is more pronounced in autarky, where alternative energy represents 72% of the energy profile, with the share of oil being the smallest amongst scenarios.

The climate policy affects overall direct economic effects and other general equilibrium impacts, assessed here by welfare and GDP. Welfare is a macroeconomic indicator to express the level of prosperity of economic agents. In EPPA6, it is measured based on variations in consumption levels, which translate both income and relative price changes of the representative consumer, as an indicator for the induced change in utility. Additional economic cost of the sectoral ETS trading is evaluated in relation to impacts on GDP, that encompasses directly net export value (exports minus imports) and investments. These macroeconomic results are reported as percentage changes between policy scenarios and the No-Policy scenario in Figure 24. Additionally, these endogenous variables along with carbon price are included in the sensitivity analysis for the standard linkage scenario (Bra-EU-Trade), included in Appendix E.

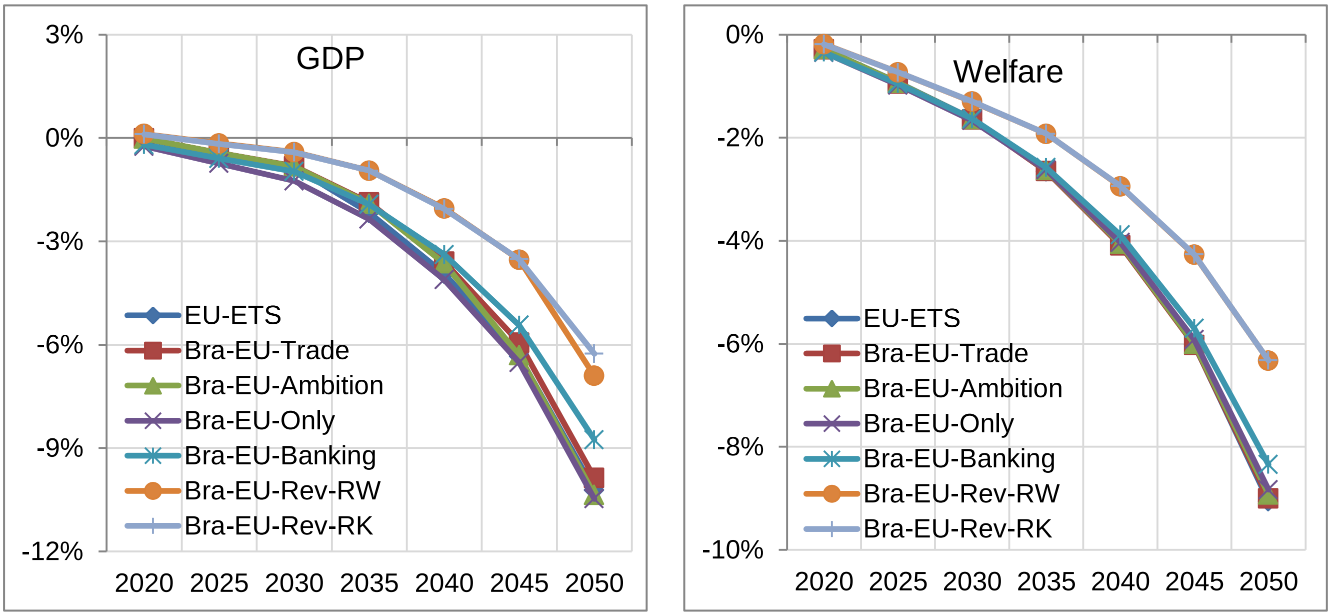
According to Figure 24, sharing the carbon constraint improves GDP and welfare in Brazil since it lowers the cost of the policy domestically and hence, the price to be paid by the economic agents. However, welfare reductions are lower than GDP's, although losses are very similar among the simulated scenarios. Changes in consumption are comprised in the GDP loss, as well as changes in investment levels, international trade and government expenditures. In the 2020-2030 horizons, the Bra-ETS presents the deepest decline of GDP and welfare, approximately 4.2% and 2.5%, respectively. This occurs due to the fact that the covered sectors face higher abatement costs as a result of the deep mitigation assumed for Brazil by 2030. Moreover, the electricity sector is relatively low-carbon, and as a result has limited opportunities to cut emissions.

Conversely, if the link is agreed so that Brazil assumes a lower level of mitigation, economic costs drop and welfare losses are the smallest by 2030. Yet mitigation costs are the lowest when carbon revenues are reinvested back into alternative technologies, namely the production of renewable energy in the Brazilian economy. In the long term, GDP losses range between 2.7% and 5.5%, being less negatively impacted in the Bra-EU-Ambition and Bra-EU-Rev-RW scenarios by 2050. Compared to No-Policy levels, welfare is highly impacted by a domestic ETS or a linked system in which allowances can be banked over periods. Again, whether or not other regions commit via ETS, a linkage with basic ETS design yields the same costs for the economy and society to meet the mitigation target in the long term. However, if Europe agrees with a lower ambition, Brazil benefits by a 0.4% gain in welfare.

Figure 24 - Changes in GDP and welfare in relation to the No-Policy scenario in (a) Brazil and (b) Europe



(a)



(b)

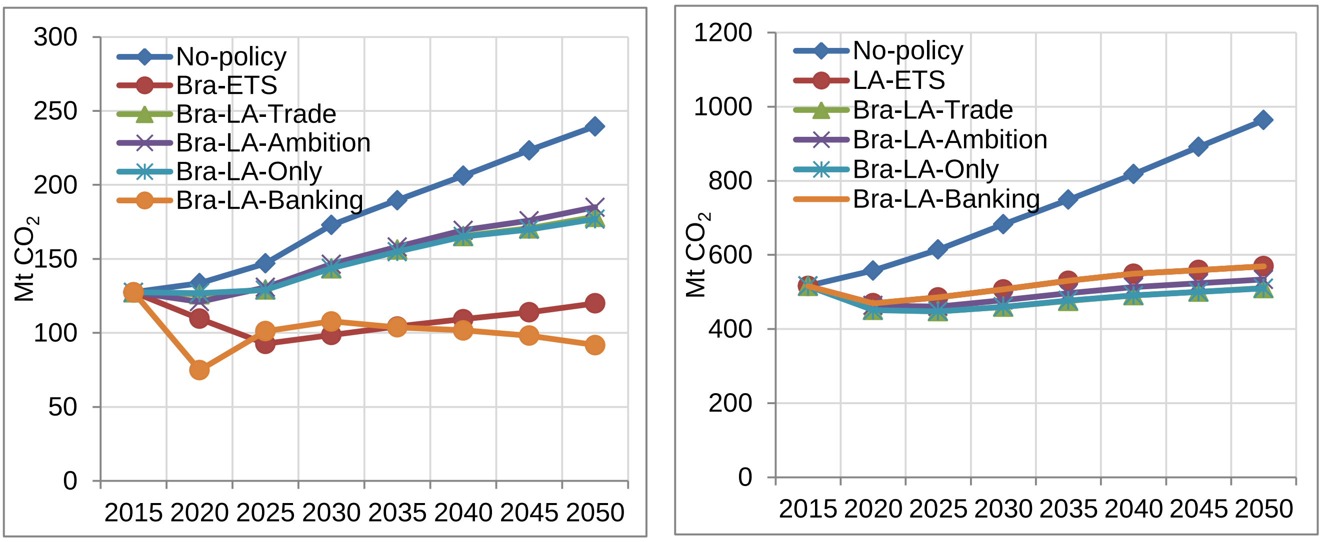
The impacts on GDP and welfare in Europe are very similar between the scenarios simulated, induced either by the related financial transfers and the reduction in aggregate output as a result of the general equilibrium effect. The decline in GDP is slight by 2030 and gradually gets more pronounced as the cap is tightened, with the more severe reduction being approximately 10%, i.e. where further effort to abate emissions is required such as if no bilateral trading is allowed or in the Bra-EU-Ambition and Bra-EU-Only situation in 2050. Welfare implications follow the same downward trend over the period, ranging from 0.2% to 9.1% compared to the No-Policy scenario. However, where revenues are recycled back to Brazil, irrespective of how, the linkage improves these negative economic effects.

In short, evidences suggest that the linking of an emerging sectoral ETS from a developing country, such as Brazil, to an established scheme such as the EU ETS, can to a certain extent lead to welfare benefits for the jurisdictions involved. The linkage modelled in this thesis underlines that an international ETS that recycles revenue from trading towards renewable energy production (including by subsidising capital) would be the most cost-effective option in terms of economic performance and effects on welfare. One alternative to improve economic performance for Brazil could be to agree on a trade deal where it adopts less ambitious mitigation goals as it minimises distributional effects from linking under the policy framework modelled.

5.3.2 A regional ETS cooperation: linking Brazil and Latin America

An integrated emissions trading system in Latin America could be a relevant strategy to provide adequate price signals for a low-carbon region, as well as the need to participate in global climate efforts. Overall, the region is responsible for approximately 6% of total global emissions, whereas emissions from covered sectors represent on average 3% of global electricity and energy-intensive emissions. The trajectory of these sectoral emissions from 2020-2050 is displayed in Figure 25 considering different ETS designs and in Figure 26 for different methods of recycling the trading revenue. The associated carbon price is presented in Figure 27.

Figure 25 - ETS emissions for Brazil (a) and Latin America[78] (b) under different ETS design scenarios

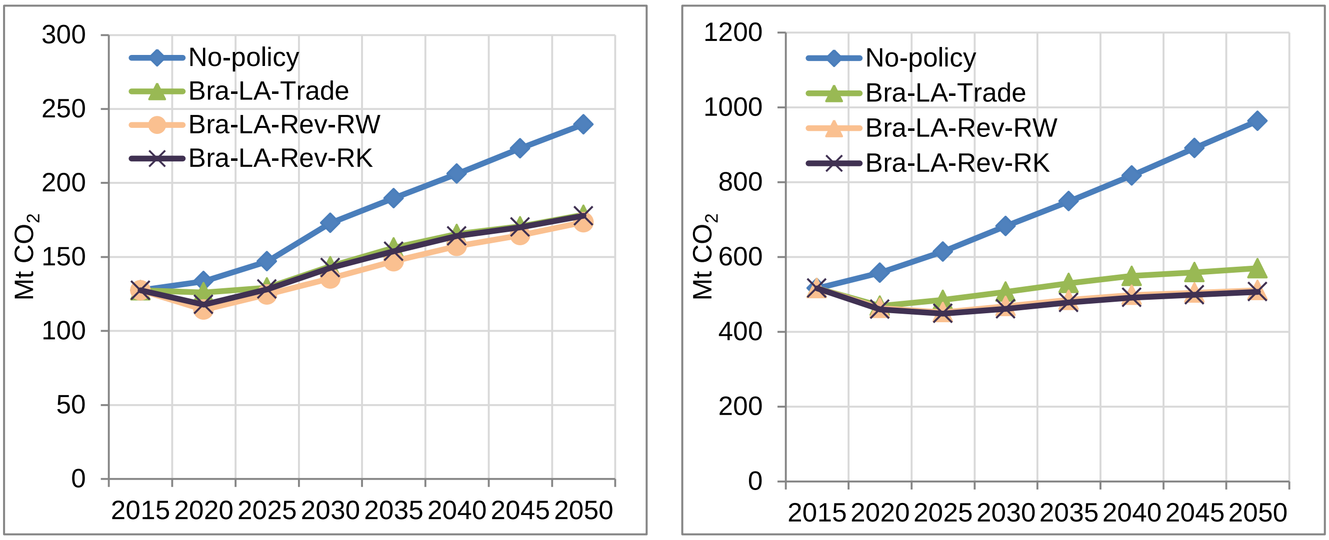


(a) Brazil (b) Latin America

In the Bra-LA-Trade scenario, Brazil is a net importer of carbon permits from the covered sectors in Latin America, with a transfer of 45 and 58 million tonnes of CO_2 in 2030 and 2050, respectively. In comparison to the No-Policy scenario, this amount represents an addition to the emissions permits from covered sectors in Brazil of 46%, which are purchased at US$88/t$CO_2$ in 2030.

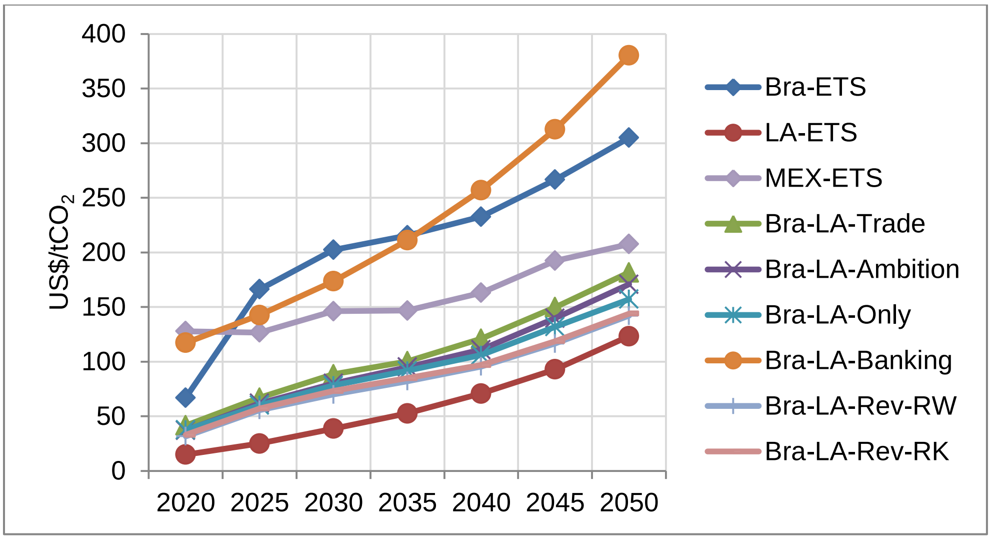
[78] Latin America refers to the rest of Latin America (LAM and MEX in EPPA6), with Brazil separated.

Figure 26 - ETS emissions for Brazil under different revenue recycling mechanisms



(a) Brazil (b) Latin America

Figure 27 – CO_2 prices under different scenarios



For Latin America, the linkage implies a reduction of 47 million tonnes from the domestic ETS situation in 2030. In the long-term, the carbon price is equalised at US$182/t$CO_2$ and reduces 59 million tonnes compared to the domestic ETS policy. Allowances cost less to Latin American covered sectors in the LA-ETS scenario, approximately US$40/t$CO_2$ in 2030. This carbon price is consistent with previous research (Kober *et al.*, 2014; Clarke *et al.*, 2016), where the costs of emitting carbon range from about US$15/t$CO_2$ to US$50/t$CO_2$ to obtain a 20% emission reduction.

Among the additional provisions incorporated into the ETS design, banking of allowances promotes further abatement by 2030 in Brazil, or alternatively, recycling revenues to the production of renewable energy. From 172 million tonnes of CO_2 in the No-Policy scenario for 2030, sectoral emissions fall to 108 and 135 million tonnes in the Bra-LA-Banking and Bra-LA-Rev-RW scenarios. In this sense, inter-temporal permit trading appears to provide a strong incentive for early action in Brazil. However, it creates scarcity in the market so that purchasing permits is very expensive in 2050. As a result, mitigation in the integrated system is limited in subsequent periods.

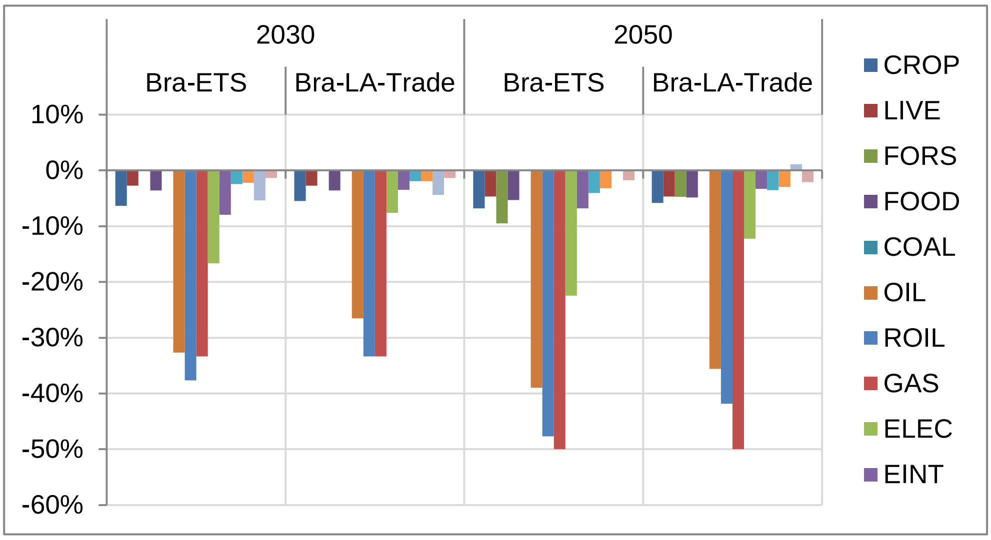
On the other hand, the minimum cost of carbon possible in a regional cooperation is achieved when revenues are redistributed to spur technological change, corresponding to US$ 70/tCO_2 and US$142/t$CO_2$ in 2030 and 2050, respectively. In a combined framework of Brazil and Latin America, aggregate ETS emissions account for 2% of global emissions and 3% of global sectoral emissions in 2030, which increase to 2.1% and 3.5%, respectively, in 2050. Compared to the Brazil-Europe linkage, where the share of global emissions diminishes, this increase occurs as consequence of being in a linkage with a relatively less ambitious target.

In terms of intra-jurisdiction mitigation, the majority of emissions reductions in both regions stem from the power sector in all scenarios, indicating that it has more alternatives for substituting fossil fuel-based energy sources with low-carbon technologies. For Brazil, the greatest mitigation level is achieved by the electricity sector in autarky or in the Bra-LA-Banking scenario, when it almost fully decarbonises. This level of decarbonisation reflects the relatively higher ambitious target that Brazil, alone, is committed to in the context of a domestic ETS.

Conversely, a linking agreement enables Brazil to have a greater aggregate cap by engaging in a link with Latin America than in the linkage with Europe. As outlined before, this is closely related to the level of ambition admitted to Brazil and the trading partner, which tend to benefit economically from the reduced costs to mitigate by trading permits. In the Brazil-Latin America link, Brazil obtains cheaper permits than linking with Europe. It is worth noting that banking allowances for future usage in the regional linkage generates the highest carbon price and thus, provides a stimulus to cut emissions throughout the period of compliance for Brazil. Due to the abatement costs faced by energy-intensive industries, it is preferable to acquire permits in the market in a linkage situation thereby reducing only 16% of their emissions with the inclusion of banking provisions, or 6% in other linking scenarios.

For Latin America, intra-jurisdiction mitigation is the lowest in the LA-ETS and Bra-LA-Banking scenarios, which provides flexibility to regulated sectors to cut emissions. However, in 2030 an increase of 4% in emissions of energy-intensive industries is observed when revenues are used to subsidise labour in all sectors of the economy. Even though most of the mitigation is driven by the Latin American electricity sector, the gap compared to mitigation of energy intensive industries is lower than in the Brazilian case.

Figure 28 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-LA-Trade linking scenario for Brazil

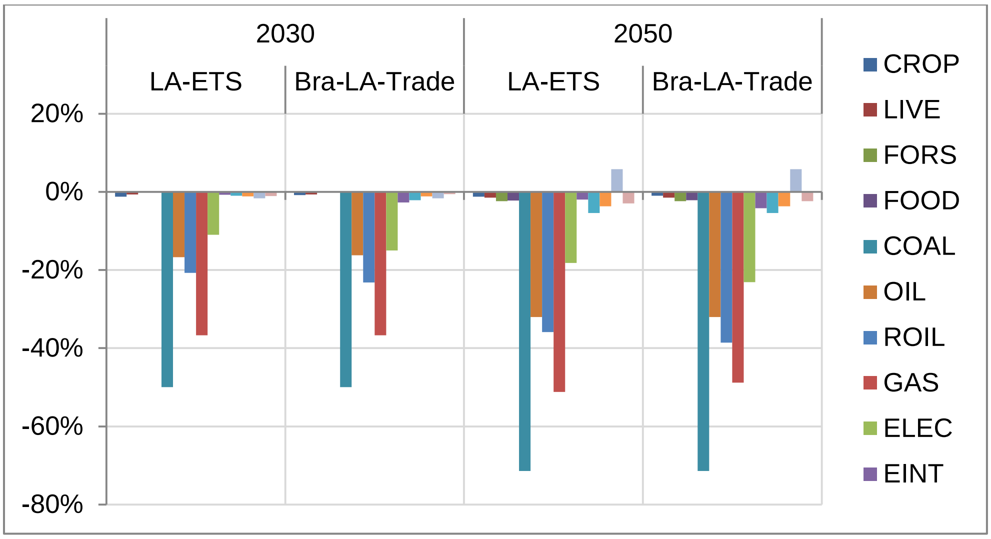


The activity level of energy-intensive sectors in Brazil is less affected by the ETS, as illustrated in Figure 28. Linking mitigation strategies with Latin America implies lower costs for both ETS and non-ETS sectors than linking with

Europe. However, losses in output are significant for refined oil, oil and gas sectors, which are highly impacted by the carbon costs.

According to Figure 29, sectoral output changes from the No-Policy scenario reveal a significant impact on non-ETS sectors in Latin America, in particular for the coal and gas, but also for the oil and refined oil sectors. For coal, production becomes more expensive and the same impact is verified regardless of linking with Brazil over the period. Interestingly, if sectoral trading is permitted, the activity level of refined oil and electricity decreases more than the domestic policy, since opportunities of abatement are lower.

Figure 29 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-LA-Trade linking scenario for Latin America



International trade of domestic goods from energy-intensive sectors is also impacted by the climate policy, as depicted in Table 16. The negative effects on the volume of exports are greater in bilateral trade with Europe. In this case, there is a decline in exports, especially in autarky where domestic goods are more expensive than foreign goods. Latin America is an important trade partner of the Brazilian energy-intensive industries. A linkage between the two regions reduces the losses in competitiveness for the sector which is, in turn, only slightly affected.

Financial transfers of trading from Brazil are presented in Table 17 below. Positive values translate the amount paid by Brazil to Latin America, whereas for negative values the direction of payments is the inverse. Two effects are highlighted: intra and inter flow of permits. As the price increases along the years, there is also a growth in the cost of carbon for the covered sectors, as well as the volume of permits flowing between the two regions. In Bra-LA-Trade in 2030, the volume of permits traded is 45 million tonnes of CO_2, increasing to 57 million tonnes in 2050, corresponding to US$4 and US$ 10.6 billion, respectively.

Table 16 – Changes in the volume of production exported from Brazilian energy-intensive sector to other regions compared to No-Policy scenario

Importers	2030		2050	
	Bra-ETS	Bra-LA-Trade	Bra-ETS	Bra-LA-Trade
AFR	-5%	-1%	-6%	-1%
ANZ	-1%	0%	-1%	0%
ASI	-4%	-1%	-5%	-1%
CAN	-3%	-1%	-4%	-1%
CHN	-5%	-1%	-6%	-1%
EUR	**-18%**	**-5%**	**-14%**	**-3%**
IDZ	0%	0%	-1%	0%
IND	-1%	0%	-1%	0%
JPN	-5%	-1%	-5%	-1%
KOR	-2%	-1%	-3%	-1%
LAM	-20%	-6%	-15%	-3%
MES	-3%	-1%	-4%	-1%
MEX	-4%	-1%	-5%	-1%
REA	0%	0%	-1%	0%
ROE	-1%	0%	-1%	0%
RUS	0%	0%	-1%	0%
USA	-17%	-5%	-14%	-3%

Table 17 – Total financial transfers of CO_2 permits from Brazil (in 2007 US$ billion)

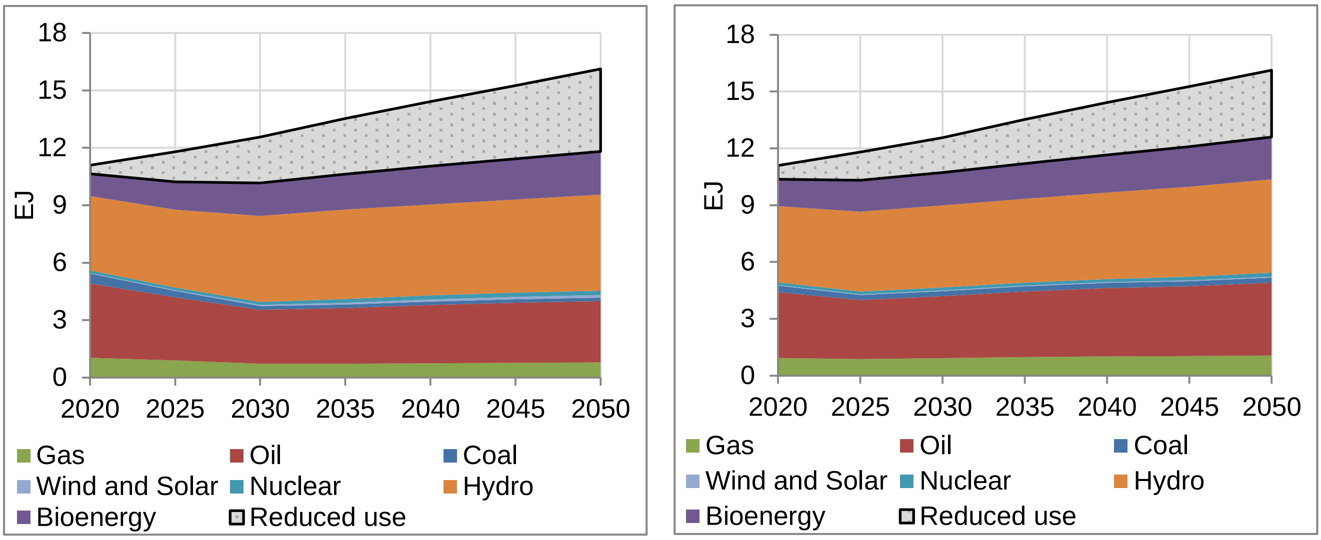
Scenarios	Bra-LA-Trade	Bra-LA-Ambition	Bra-LA-Only	Bra-LA-Banking	Bra-LA-Rev-RW	Bra-LA-Rev-RK
2020	0.7	0.4	0.7	-4.1	0.2	0.3
2025	2.5	2.4	2.2	1.2	1.8	2.0
2030	4.0	3.8	3.5	1.6	2.6	3.2
2035	5.2	5.1	4.6	-0.1	3.5	4.2
2040	6.8	6.7	5.9	-1.9	4.6	5.3
2045	8.5	8.7	7.4	-4.9	5.9	6.7
2050	10.6	11.1	8.9	-10.7	7.6	8.3

If linked to a developing region sectoral ETS, considering the policy targets simulated here, Brazil also becomes a buyer of emissions. The only exception is when the ETS is designed to allow for banking of permits across periods. In this case, there are monetary transfers to Brazil from 2035 onwards, totalling around US$10.7 billion in 2050. When joining Brazil in the designed ETS, Latin America is encouraged to promote greater mitigation and sell permits. Thus, whilst achieving further emissions reductions in Brazil goes along with increasing costs, Latin America's covered sectors benefit by obtaining income from Brazil. These findings demonstrate a reallocation of emissions and financial transfers among sectors and participants of the linked system, i.e., emissions reductions and carbon permits to a larger extent from electricity sector are transferred to Brazil.

At the inter-jurisdiction level, the Brazilian electricity sector purchases more permits from Latin America than energy-intensive industries, unless revenues are recycled to the production of renewables, or if banking provisions are introduced. Under these scenarios, the demand for permits from the Brazilian electricity sector is reduced. A comparable trend is observed in the supply side, where the greater abatement in the Latin American electricity sector generates US$2.8 billion in carbon permits to be exported in the Bra-LA-Trade scenario in 2030 and US$ 7 billion in 2050.

A key objective of a sectoral ETS proposal is to stimulate investments in low-carbon technologies. By regulating the electricity sector there is an increase in the price of electricity, which changes relative costs of different energy sources. Linking sectoral ETS systems in Brazil and Latin America does not have significant effects on total energy consumption for Brazil compared to a domestic ETS. Contrariwise, this sectoral trading would partially reverse the changes obtained in the domestic ETS. In this case, Figure 30 suggests a more important role for low-carbon technologies relative to fossil fuel-based primary energy. As a result, reductions in the total energy use from the No-Policy scenario in 2020-2050 are more evident with a domestic ETS, corresponding to 17 EJ. Alternative energy comprises 64% of the energy mix as opposed to 58% in the standard Brazil-Latin America trading scenario.

Figure 30 – Primary energy use in Brazil: Domestic ETS vs. Linkage (Bra-LA-Trade)

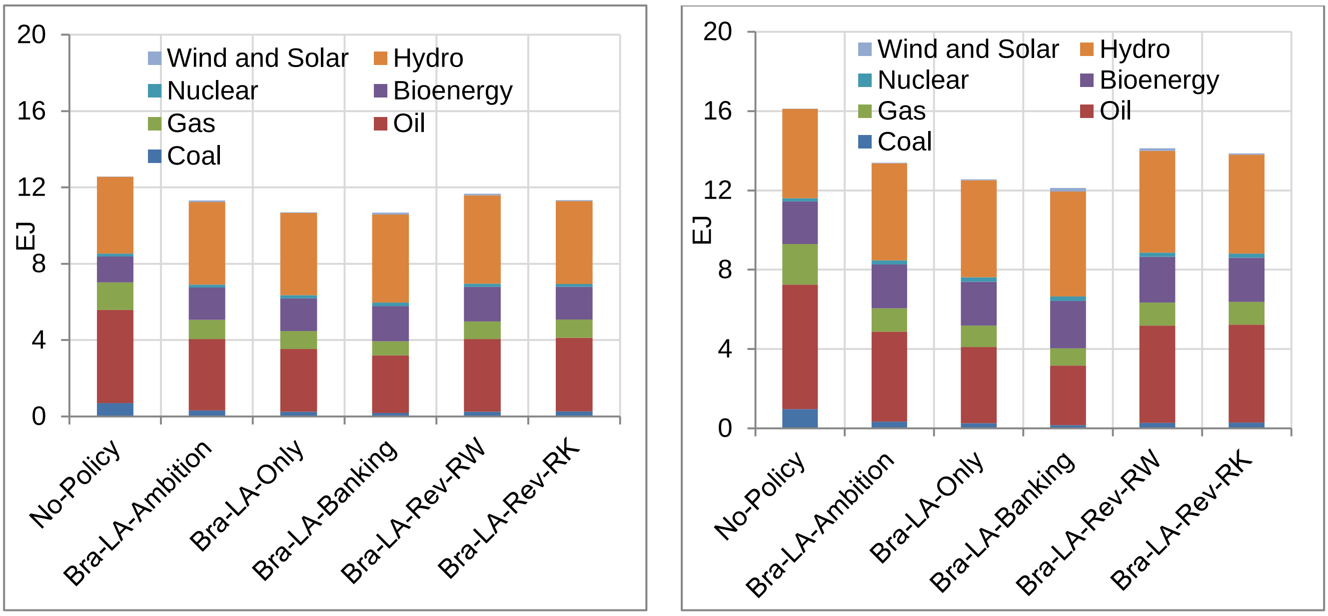


(a) Bra-ETS (b) Bra-LA-Trade

Under different ETS design scenarios, the fossil energy consumption of Brazil varies, as depicted in Figure 31. For example, in the Bra-LA-Ambition scenario for 2030, the share of carbon-intensive sources is 45%, where the use of coal, oil and gas is 0.3 EJ, 3.7 EJ and 1 EJ, respectively. In comparison, only 3.9 EJ of conventional technologies are consumed in the Bra-LA-Banking scenario in 2030. This is understandable as the degree of stringency for Brazil is high in the ETS design with banking, exerting greater pressure on the energy structure and resulting in positive effects on the use of bioenergy, nuclear and renewables. This is the scenario that enhances energy substitution the most in Brazil.

Examining the consumption of oil, the predominant fossil fuel used in Brazil, a decrease of 22% from the No-Policy scenario in 2030 and 2050 is observed, if revenues are allocated into the use of capital or the production of renewables. This is in contrast to a 38% and 52% reduction for the same time periods in the Bra-LA-Banking scenario. Simultaneously, the Bra-LA-Rev-RW scenario is the second-best sectoral trading option for enhancing the share of alternative energy in Brazil. Yet, hydroelectricity consumption prevails. Since the expansion of hydropower may be limited, relying on other low-carbon technologies is fundamental to Brazil.

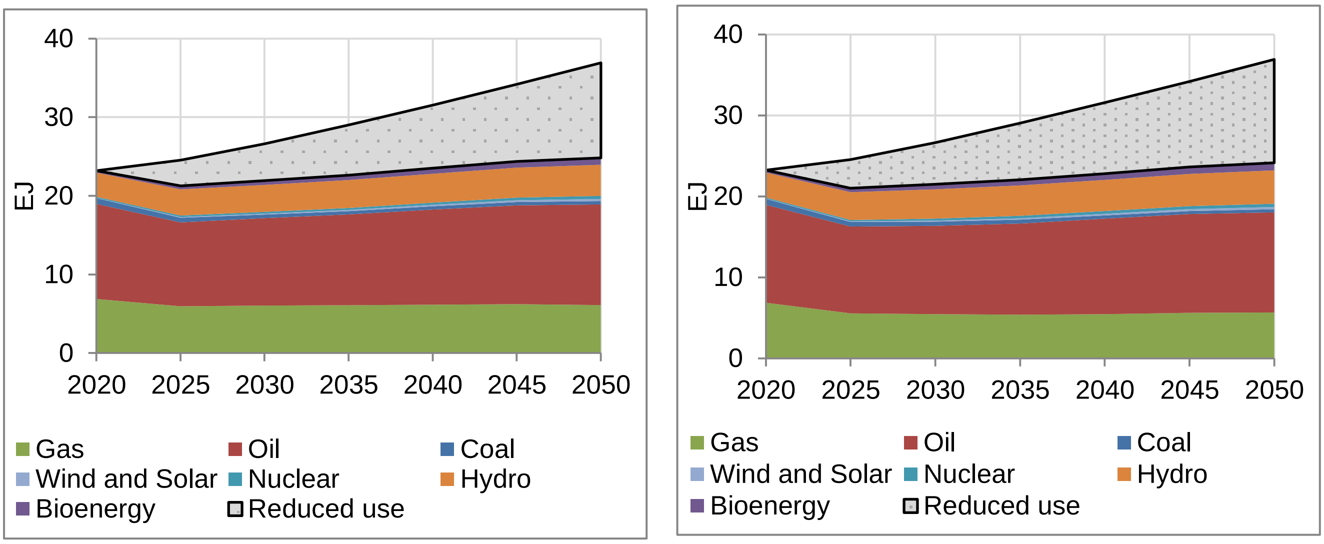
Figure 31 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for Brazil



(a) 2030 (b) 2050

Due to differences in abatement costs as well as the composition of the energy mix, the impacts of a sectoral ETS linkage differ between Brazil and Latin America. A sectoral ETS modifies the energy profile of Latin America by inducing larger consumption of low-carbon technologies compared to a domestic policy. Such change results from the fact that sectoral trading restricts more emissions from domestic sources than in the LA-ETS. For comparability, Figure 32 presents the results for the domestic and the standard linkage scenarios, where the latter reduces 48 EJ over the period compared to the No-Policy scenario. In the linking scenario, the total energy use is 21.5 EJ and 24.2 EJ in 2030 and 2050, respectively. These correspond to 10.9 EJ, 5.6 EJ and 3.6 EJ of oil, gas and hydro technologies in 2030, and 12.4 EJ, 5.7 EJ and 4.1 EJ respectively in 2050.

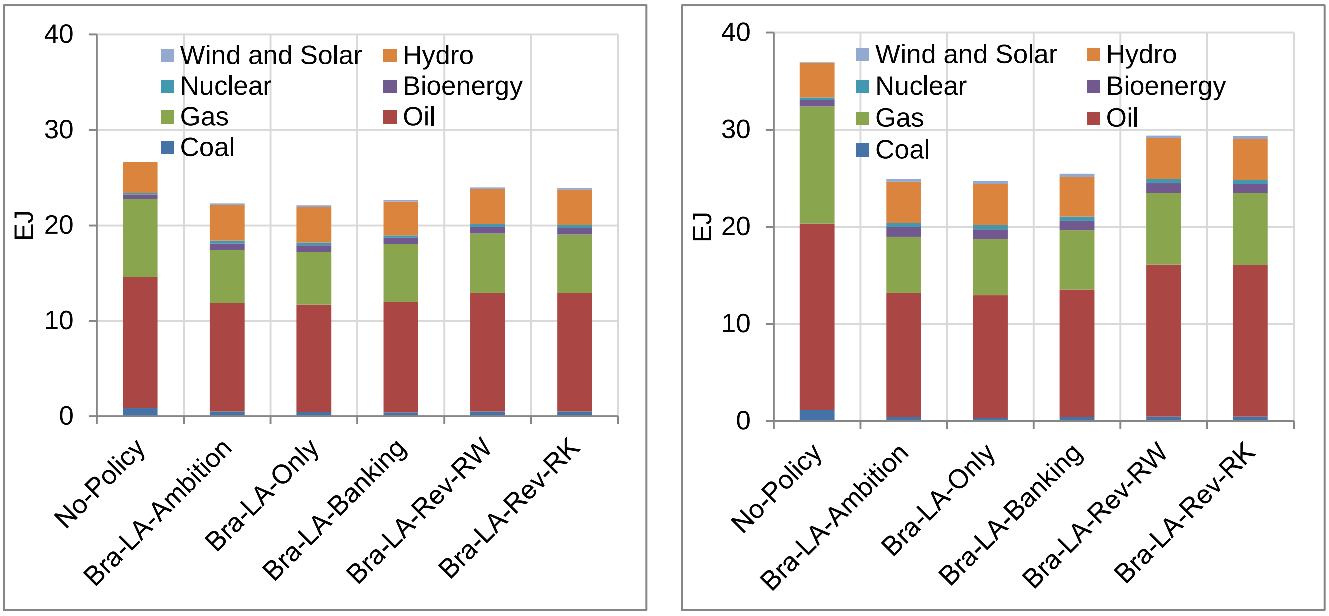
Figure 32 – Primary energy use in Latin America: Domestic ETS vs. Linkage (Bra-LA-Trade)



(a) LA-ETS

(b) Bra-LA-Trade

Figure 33 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for Latin America



(a) 2030 (b) 2050

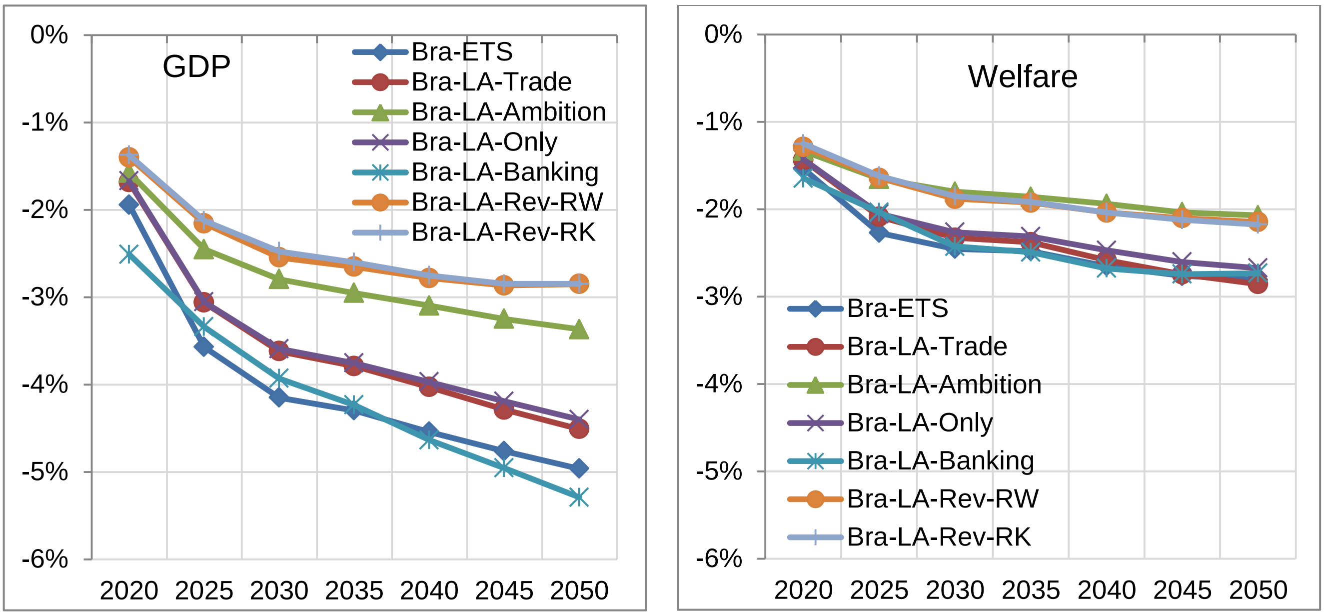
Within a linked framework, Latin America uses on average 23.2 EJ, out of which 17.4 EJ is carbon-intensive energy. Results in Figure 33 reveal that the consumption of oil, gas and hydroelectricity remains predominant in the energy mix. However, in comparison to the No-Policy scenario, there is a reduction of approximately 40% and 60% of coal consumption in all trading scenarios in 2030 and 2050 respectively. Where revenues are reintroduced, reliance on the carbon-based energy source continues, in particular for oil and gas.

For that purpose, if the ETS is active only in Brazil and Latin America or if Brazil adopts a less strict cap in the linkage, replacement is greater as there are incentives in Latin America to invest in technological improvements in order to abate more and sell permits to Brazil. The Bra-LA-Trade scenario also provides a strong and lasting carbon price signal to achieve the required transition to a low-carbon economy, which involves higher infrastructural investments and thus, larger costs. As for the use of alternative energy, the Bra-LA-Only scenario induce the use of bioenergy and nuclear, whereas the Bra-LA-Banking diminishes their use, as well as the share of hydroelectricity. In fact, the effect of the carbon constraint on the consumption of hydro, wind and solar is not linked to the ETS design.

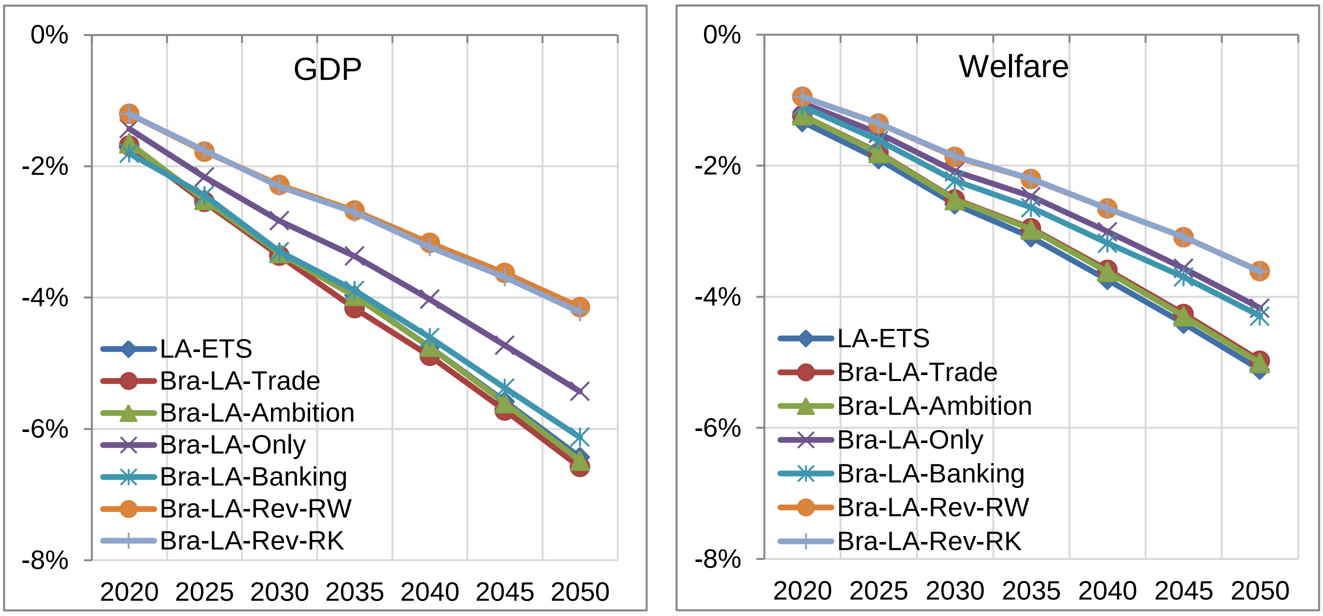
Among potential trading partners under investigation, Latin America is the most homogenous compared to Brazil in terms of macroeconomic and energy profile. In accordance with theory, a regional sectoral ETS linkage can reduce the adverse impacts of the carbon constraint on the economy with higher overall costs, namely Brazil. Figure 34 shows results for GDP and welfare, which reflect the macroeconomic and other general equilibrium effects of reducing carbon emissions in response to the carbon price signal. Notwithstanding this, results are linked to the representation of the world in the model, which disregards complexities and uncertainties of implementing a sectoral ETS linkage. In an attempt to confirm whether evidences are robust, Appendix E reports results for GDP and welfare, as well as carbon price under different growth and energy efficiency assumptions for the Bra-LA-Trade scenario.

The policy-induced GDP for Brazil in 2030 is on average 3% lower than the No-Policy scenario, as opposed to 4.2% in the Bra-ETS. As a result of the general equilibrium effect, there are negative impacts on GDP and welfare in Latin America. This is consistent with other studies analysing the macroeconomic impacts of climate mitigation in Latin America (Tavoni and Socolow, 2013; Bowen, Campiglio and Tavoni, 2014; Kober *et al.*, 2014), where both GDP and welfare is expected to range from -6% to +1% depending on the proposed targets and the resulting carbon price regime. When comparing the developing regions investigated, results show that the effect on Brazil has similar magnitude as in Latin America, but the former experiences the deepest drop in GDP and welfare in autarky. Hence, sharing the carbon constraint improves GDP and welfare in Brazil since it lowers the cost of the policy domestically and hence, the price to be paid by the economic agents.

Figure 34 - Changes in GDP and welfare in relation to the No-Policy scenario in (a) Brazil and (b) Latin America



(a) Brazil



(b) Latin America

For both Brazil and Latin America, the less costly option in terms of ETS design is to include mechanisms for recycling revenues to the production of renewables, as a compensation for the competitiveness losses and stimulus for technological substitution in Brazil. Otherwise, trading emissions permits does not reduce the burden of a sectoral constraint in Latin America, compared to a domestic policy. Results indicate these scenarios as the most appropriate for alleviating negative effects on welfare. To the extent that carbon costs induce an overall rise in electricity prices, which is detrimental to consumers, both GDP and welfare losses can be more accentuated if allowances are distributed to covered entities via auctioning.

In short, evidences suggest that a regional integration through a sectoral ETS linkage expands opportunities for joint emissions abatement at lower costs. Considering the global climate perspective, the Brazil-Latin America linkage promotes very limited aggregated reductions in light of the limited volume of aggregated emissions being covered compared to other potential partners.

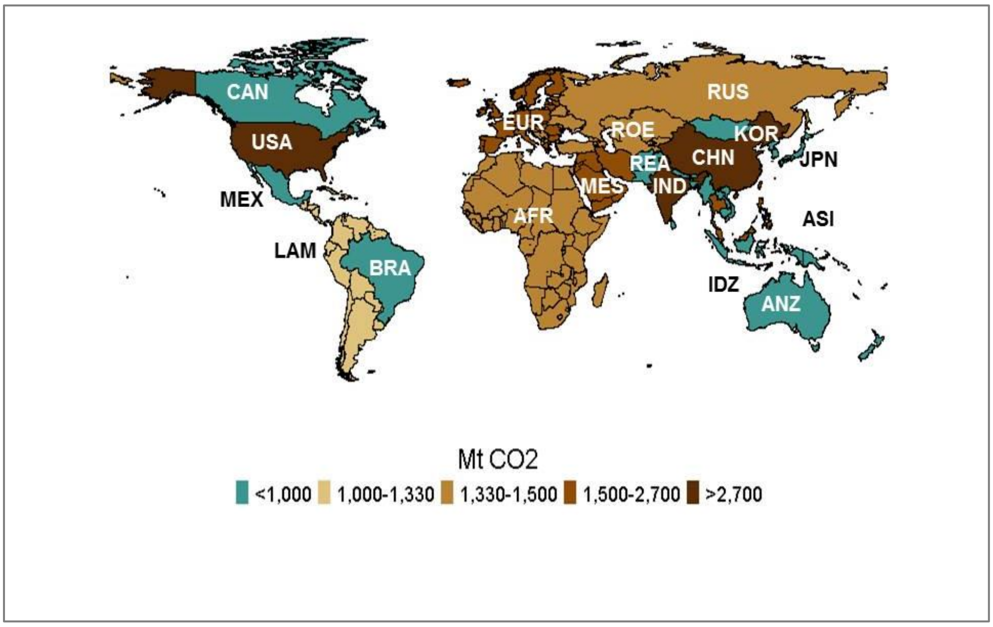
From an economic perspective, there are benefits of linking compared to a domestic ETS policy. This is also the most appropriate mitigation strategy for Brazil since it allows the energy adjustment cost to be lowered by purchasing permits. Nevertheless, negotiation of this climate agreement would be very challenging. Firstly because depending on the ETS design, countries are affected differently. For example, the use of banking pressure in Brazil to find alternatives, ultimately boosting low-carbon technologies and deeper mitigation, which generates additional income in the long-term. With banking, there are higher costs in initial years due to the anticipation of abatement, which tends to diminish over the period. However, anticipated mitigation via banking has the potential to compromise investment and capital accumulation in the economy and therefore impact long-term economic growth. In this case, policy costs in the future are not much lower than non-banking scenarios.

Despite the relatively homogenous macroeconomic and energy profile, divergences in economic and environmental interests may emerge. Hence, the most suitable ETS design has to consider the minimum costs to enable economic development to occur whilst signalling the commitment to address climate change. A trade deal that recycles revenues from trading into renewable energy production would be the most cost-effective option in terms of economic performance for both regions and modest mitigation is achieved.

5.3.3 A Brazil-China sectoral ETS linkage

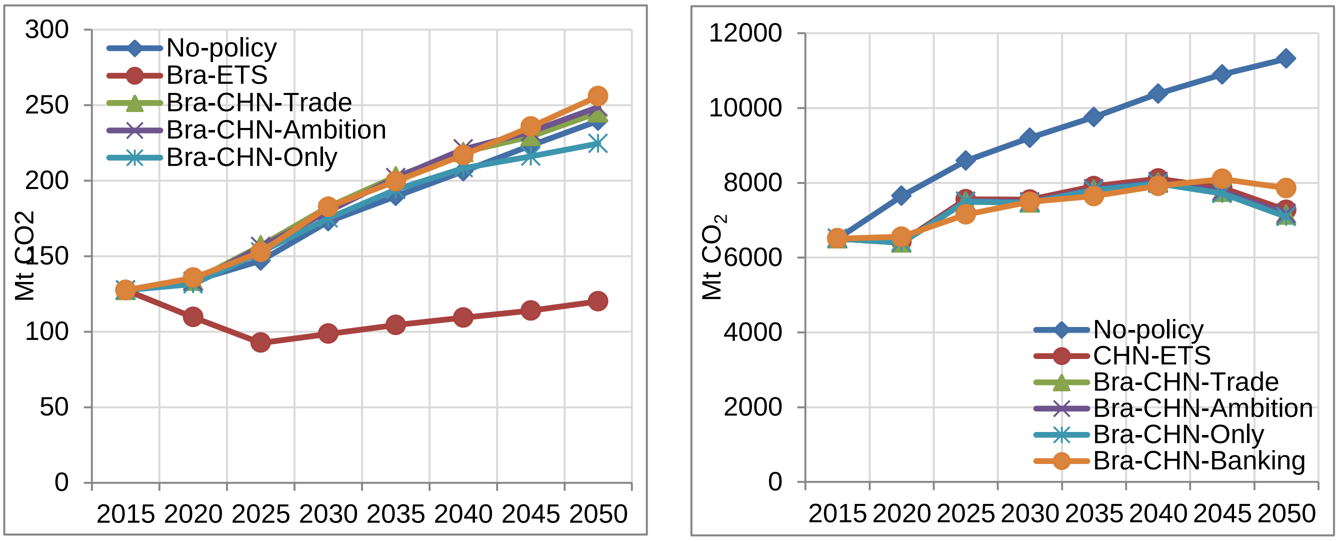
Recently, China has launched the nationwide sectoral ETS. In this modelling exercise, the proposed ETS regulates electricity and energy-intensive industries to closely align to the EU ETS setting. In the meanwhile however, China intends to cap only emissions from the power sector. In fact, China is the largest emitter in the world, as shown in Figure 35, with approximately 30% of total global emissions while Brazil is responsible for only 2%.

Figure 35 – Total CO_2 emissions in 2030



Sectoral emissions under different scenarios and carbon prices are exhibited in Figure 36, Figure 37 and Figure 38. In a global perspective, a sectoral ETS linkage between Brazil and China corresponds to 38.6% of emissions from electricity and energy-intensive sectors in 2030 and declines to 37.7% in 2050. Rather than modifying the aggregate volume of emissions compared to a domestic ETS, linkage allows both regions to negotiate permits in a way to seize abatement opportunities among them. On one hand, every attempt to avoid a climate that is 2ºC warmer must involve China, whilst on the other, binding commitments are still modest. Thus, linking to a small ETS market, such as the Bra-ETS, neither substantially increases liquidity in the carbon market, nor does it have a perceptible abatement effect compared to the implementation of a national ETS.

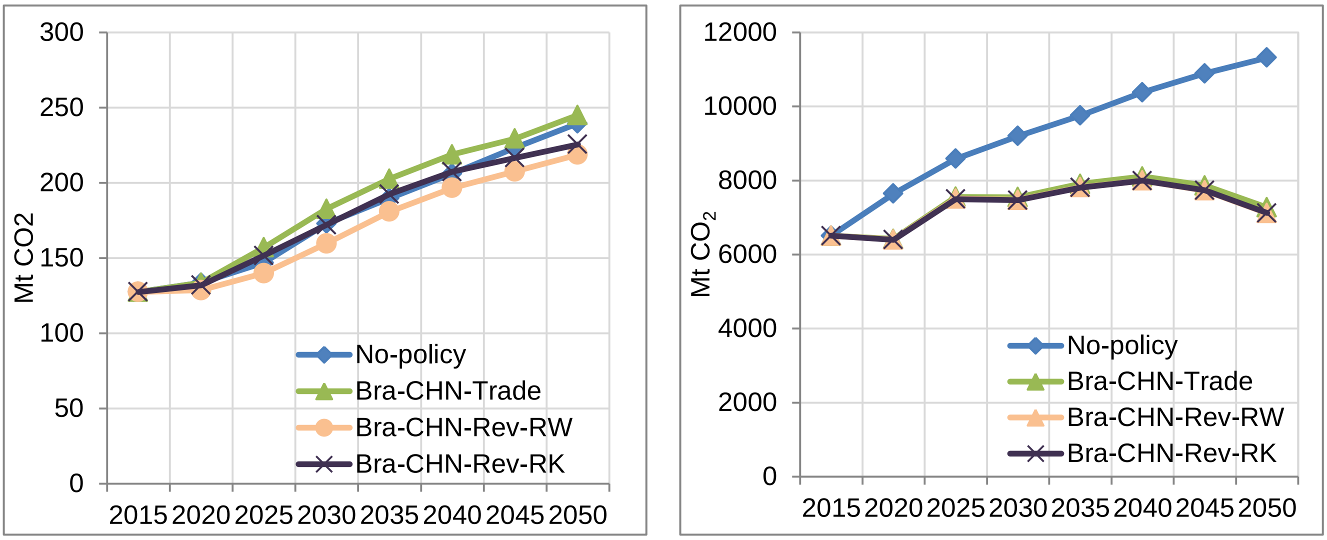
Figure 36 - ETS emissions for Brazil under different ETS design scenarios



(a) Brazil (b) China

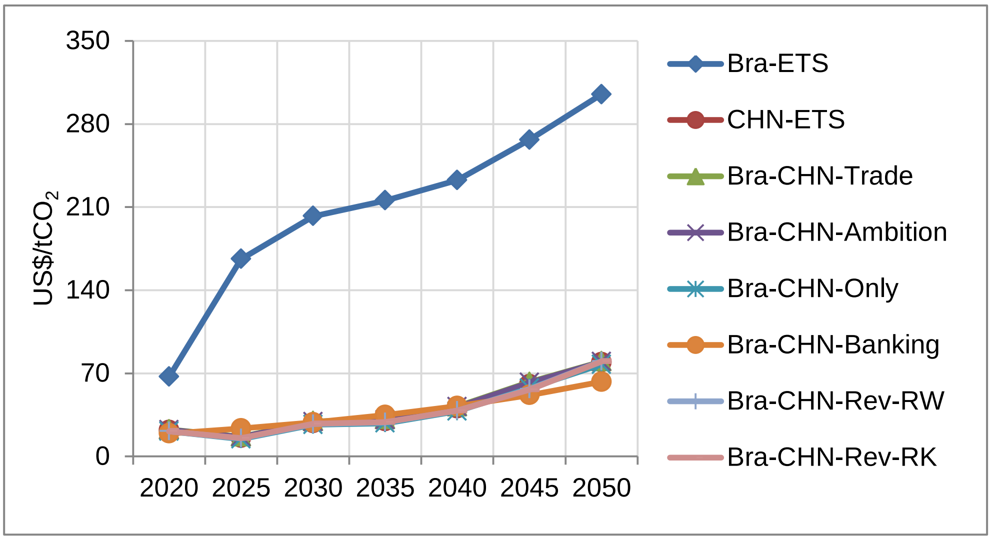
If China sets a cap on electricity and energy-intensive industries and does not trade carbon permits abroad (CHN-ETS), ETS emissions are 7.6 and 7.3 billion tonnes of CO_2 in 2030 and 2050, respectively. This amount corresponds to less than 1.7 and 4.1 billion tonnes of CO_2 than the No-Policy scenario for the same periods, where carbon permits cost US\$28/tCO_2 and US\$79/tCO_2 respectively. If it couples with the Brazilian ETS, carbon price equalises across the two systems at US\$30/tCO_2 in 2030 and US\$80/tCO_2 in 2050 in the standard Bra-CHN-Trade scenario, being almost pegged to China's autarky price. A developed-developing country linkage with China in Gavard, Winchester and Paltsev (2016) generates a lower carbon price, ranging from US\$17/tCO_2 to US\$ 24/tCO_2 in 2030.

Figure 37 - ETS emissions for Brazil (a) and China (b) under different revenue recycling mechanisms



(a) Brazil (b) China

Figure 38 – CO_2 prices under different scenarios



In fact, the cost of carbon is similar in all scenarios, thereby implying that more than the ETS design itself, the stringency of targets as well as the level of heterogeneity in energy and emissions pattern play a major role in determining demand and supply of permits in this proposed linkage, particularly because linking to Brazil provokes the same effect as the domestic ETS in terms of emissions reductions and carbon prices.

Hence, Brazil is the one that largely benefits from the linkage with China since the cost of carbon is reduced at the targeted level of emissions. This is understandable as the counterfactual carbon price for Brazil amounts to US$202 per tonne of CO_2 in 2030 and US$305 in 2050, which is higher than the Chinese ETS. Additionally, the limit on sectoral emissions is greater without sectoral trading, which is 74 and 120 million tonnes of CO_2 less than the No-Policy scenario in 2030 and 2050.

Under the linkage, there is a reverse effect on Brazil, that is, a rise in CO_2 emissions relative to No-Policy projections, notably in the electricity sector with an average of a 25% increase. This raise in sectoral emissions suggests that the Brazilian electricity and energy-intensive sectors obtain a comparative advantage compared to other sectors when linking to China. The reason for that is the lower carbon cost for sectors participating in the linkage as opposed to the sectoral taxes on non-ETS sectors, which do not have flexibility to purchase permits from other sectors or regions. By linking, the Brazilian electricity sectors increases the use of fossil fuels compared to the No-Policy scenario and/or the energy-intensive industries increase production.

Among linking scenarios, Brazil presents the greatest mitigation level if revenue recycling mechanisms are incorporated to encourage production of renewables, which reduces approximately 10% of the No-Policy emissions as opposed to 43% and 50% in Bra-ETS for 2030 and 2050. For China, joining Brazil slightly increases mitigation because it is more economically feasible to make abatement efforts to comply with the NDC targets, which are less ambitious than Brazil's. In Bra-CHN-Rev-RW, sectoral emissions total 7.5 and 7.1 billion tonnes of CO_2 in 2030 and 2050, 19% and 37% less than the No-Policy scenario, respectively.

A major concern arises from the carbon price harmonisation, particularly due to carbon price changes in the coordinated system. In view of potential competitive pressure to pass on the carbon content costs to final products, equity issues across firms, jurisdictions and income groups may emerge, affecting even those that do not participate directly in the trading. In the case of an integrated Brazil and China system, there are more abatement opportunities in the regulated sectors of China, particularly the power sector, which creates emissions permits to be supplied in the market. Demand for permits is particularly driven by the Brazilian electricity sector, which is already relatively low-carbon and to where flexibility to comply with the ETS cap is important to maintain production levels. In the long-term energy-intensive sectors are able to mitigate more than the electricity sector, around 10% compared to the No-Policy scenario.

Linking engenders, therefore, economic benefits for covered sectors in China by obtaining additional income, especially if emissions allowances were exported via auctioning. Table 18 presents the total monetary transfers for Brazil under the scenarios analysed. In Bra-CHN-Trade, the total volume of emissions imported from China is 45 and 58 million tonnes of CO_2 in 2030 and 2050, respectively. This is equivalent to US$1.3 and US$4.7 billion flowing from Brazil.

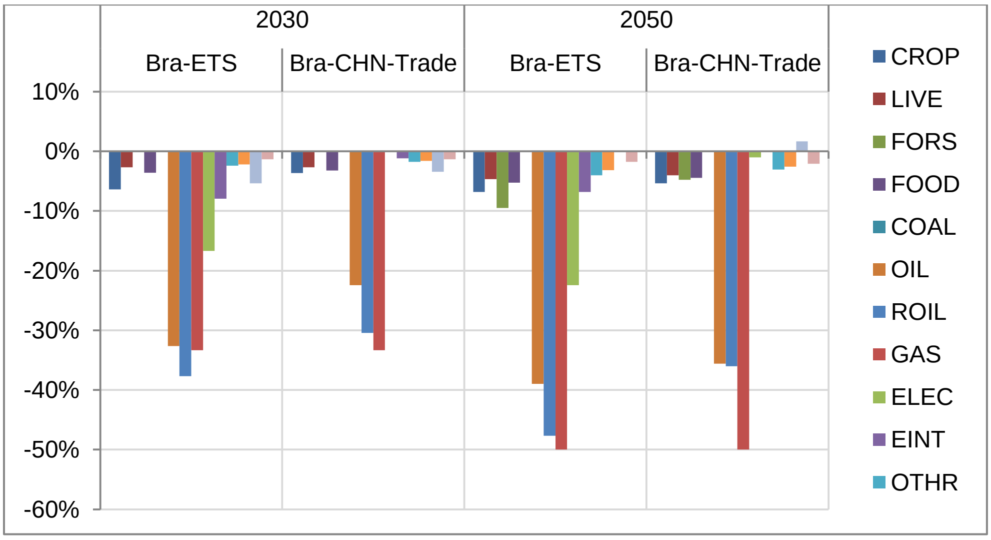
Table 18 – Total financial transfers of CO_2 permits for Brazil (in 2007 US$ billion)

Scenarios	Bra-CHN-Trade	Bra-CHN-Ambition	Bra-CHN-Only	Bra-CHN-Banking	Bra-CHN-Rev-RW	Bra-CHN-Rev-RK
2020	0.4	0.3	0.4	0.5	0.1	0.2
2025	0.6	0.6	0.5	1.4	0.5	0.5
2030	1.3	1.4	1.2	2.4	1.0	1.2
2035	1.6	1.7	1.4	3.3	1.2	1.4
2040	2.4	2.5	2.1	4.6	1.9	2.1
2045	3.5	3.9	3.2	6.3	2.9	3.2
2050	4.7	5.2	4.4	8.5	4.3	4.6

If both systems adopt a less stringent target, there is an increase in exports of permits from China, whose quantity is influenced by the size of the Chinese market and the carbon intensity of it compared to Brazil's. Similarly, the inclusion of provisions for banking in the ETS design is very costly for Brazil, which purchases US$ 8.5 billion in emissions permit by 2050. On the other hand, the use of revenues to subsidise the production of renewables in Brazil reduces the need for allowances and trading decreases. Yet, the amount transferred from China to Brazil is lowest among the investigated scenarios, demonstrating that the constraint imposed on emissions makes reductions less expensive for covered sectors comparatively.

As a result, impacts on the activity level are nullified for ETS sectors in Brazil if sectoral trading is allowed in spite of a domestic system, as reported in Figure 39. These results suggest roil is to be the energy sector in Brazil that benefits the most from linking with China, followed by the oil sector. For covered ETS sectors, it means increasing the use of this energy source if participating in an ETS linkage. Since the use of roil for electricity is generally small in Brazil, alternatively roil is able to replace electricity, gas and coal as an energy source in other sectors whereas ETS sectors use more gas and coal as they may opt to import emissions rather than further mitigate.

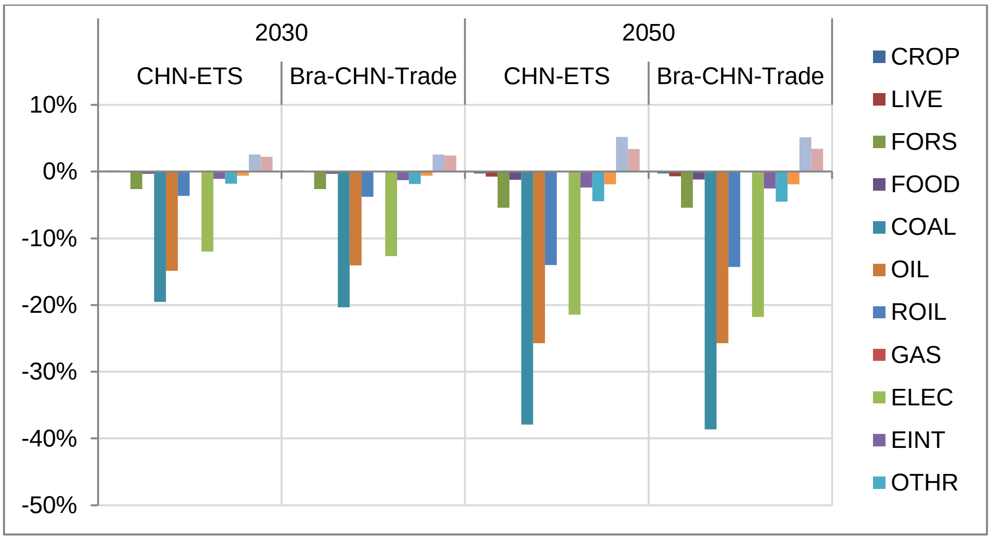
Figure 39 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-CHN-Trade linking scenario for Brazil



Although adverse effects on non-ETS sectors in Brazil exist, linking emissions mitigation strategies with China declines losses in output in some sectors, such as oil and roil. For the production of gas, regardless of linking or not, the effect of pricing the carbon is the same. This is also the case for non-

ETS sectors in China, which is depicted in Figure 40. Electricity and energy-intensive industries are slightly affected by the ETS linkage, with a reduction of only 1% in output levels.

Figure 40 – Sectoral output changes (from the No-Policy scenario) in Domestic vs. Bra-CHN-Trade linking scenario for China



At the international level, Table 19 shows that linking with the sectoral Chinese ETS represents gains in competitiveness for Brazilian energy-intensive industries in comparison to the Bra-ETS. Europe is among the most important trading partners of those sectors. By joining China, domestic goods become more competitive relative to foreign goods so that there is a small growth in imports of energy-intensive goods by Europe, Latin America, USA, Africa and China as well.

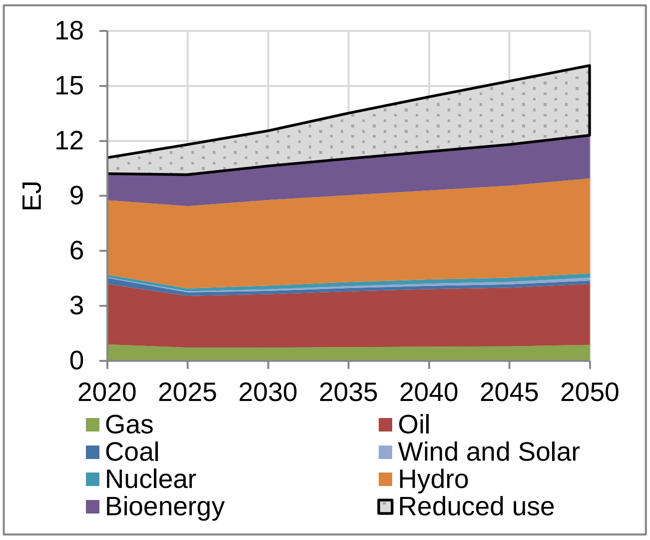
Table 19 – Changes in the volume of production exported from Brazilian energy-intensive sectors to other regions compared to No-Policy

Importers	2030		2050	
	Bra-ETS	Bra-CHN-Trade	Bra-ETS	Bra-CHN-Trade
AFR	-5%	1%	-6%	3%
ANZ	-1%	0%	-1%	0%
ASI	-4%	0%	-5%	2%
CAN	-3%	0%	-4%	2%
CHN	-5%	1%	-6%	3%
EUR	**-18%**	**2%**	**-14%**	**8%**
IDZ	0%	0%	-1%	0%
IND	-1%	0%	-1%	1%
JPN	-5%	0%	-5%	2%
KOR	-2%	0%	-3%	1%
LAM	-20%	2%	-15%	7%
MES	-3%	0%	-4%	2%
MEX	-4%	0%	-5%	2%
REA	0%	0%	-1%	0%
ROE	-1%	0%	-1%	0%
RUS	0%	0%	-1%	0%
USA	-17%	2%	-14%	6%

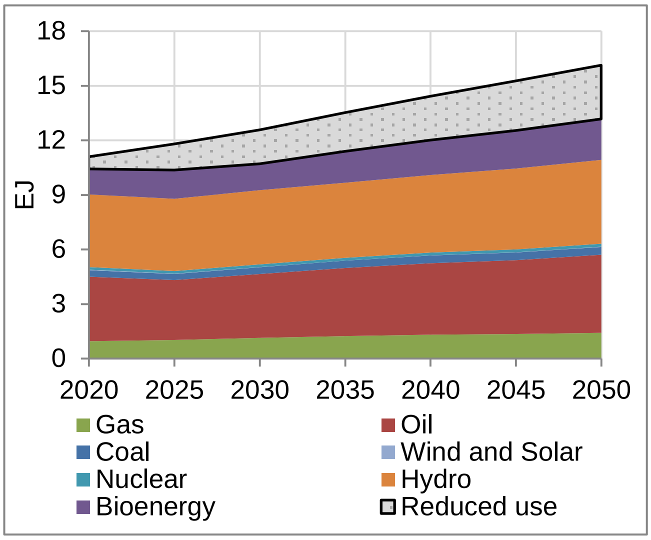
From the environmental perspective, the Brazil-China linkage could be problematic in comparison to the Brazil-Europe linkage proposal due to limited mitigation outcomes. This arises from the less ambitious Chinese ETS system, which induces Brazil to buy permits instead of curbing emissions. In this sense, the Brazil-China link could be perceived internationally as an attempt by Brazil to reduce the mitigation effort thereby benefiting from China's less stringent target. Politically, it could compromise the view that Brazil is being committed in addressing climate change.

One of the primary effects of regulating carbon emissions is to control energy sources. The modelling results for energy use capture this effect in a domestic sectoral ETS and among linked options, which is exhibited in Figure 41 and Figure 42. As outlined before, this finding is closely related to the level of ambition by Brazil and its potential trading partners, who tend to benefit economically from the reduced costs to mitigate by trading permits. Under the proposed climate agreement, linkage partially reverses technological changes induced by the domestic ETS in Brazil, where traditional sources rise from 3.8 EJ in the Bra-ETS to 5 EJ.

Figure 41 – Primary energy use in Brazil: Domestic ETS vs. Linkage (Bra-CHN-Trade)



(a) Bra-ETS



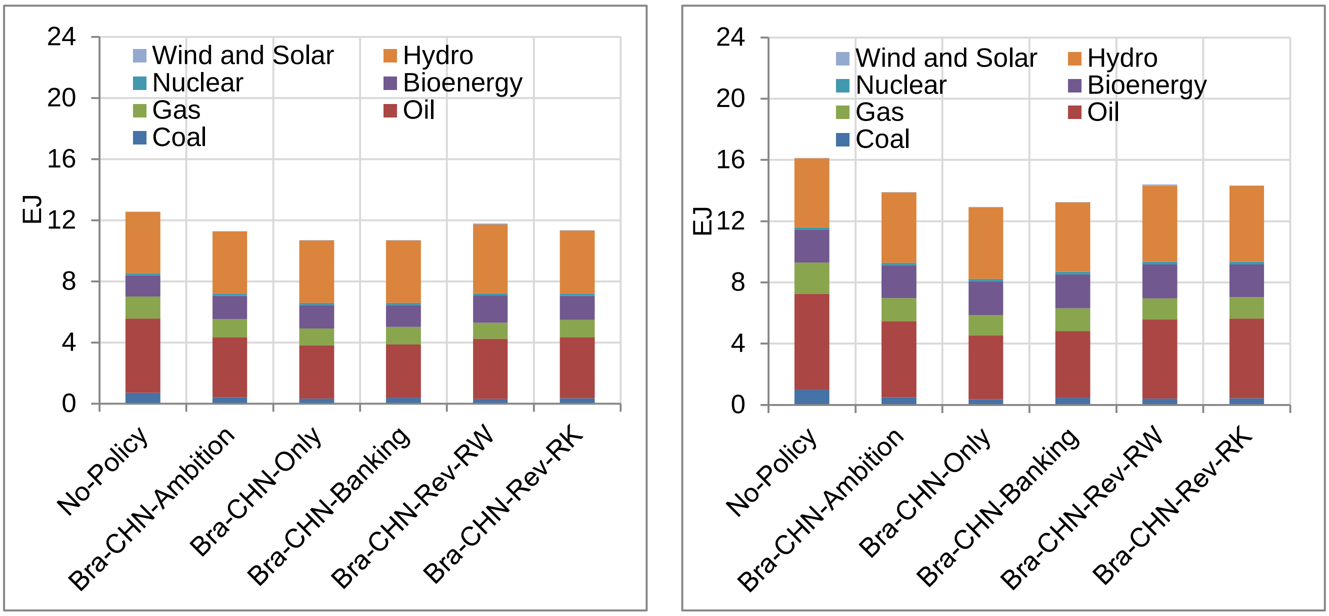
(b) Bra-CHN-Trade

In this sense, the total amount of energy reduced in Brazil from the No-Policy scenario is 17.2EJ, 3EJ more than the Bra-CHN-Trade. The share of conventional technologies is 36% in Bra-ETS and 47% in Bra-CHN-Trade, which represents a higher portion than in the No-Policy scenario, i.e. 43%. This means that linking with China tends to discourage the replacement of fossil fuel-based energy.

According to Figure 42, there are small proportional changes in some sources in Brazil while different designs for sharing the carbon price cause large variations in others. Firstly, it is observed that a less stringent carbon constraint pushes up the use of coal and gas. For example, in the Bra-CHN-Ambition scenario, consumption of coal corresponds to 127% and 167% of that in the Bra-ETS in 2030 and 2050 respectively. In line with that, the primary effect of stimulating the production of renewables is a decrease of coal, which is 29%

and 22% lower than in the case of less ambition for the same period. Even though recycling revenues for expanding the production of renewables is followed by increases in the use of oil; consumption of bioenergy, hydro and solar and wind are enhanced the most, by 4%, 10% and 9122% respectively in 2050. As for nuclear, there is no effect of linking in all scenarios in 2030. Since the expansion of hydropower may be limited, relying on other low-carbon technologies is fundamental to Brazil.

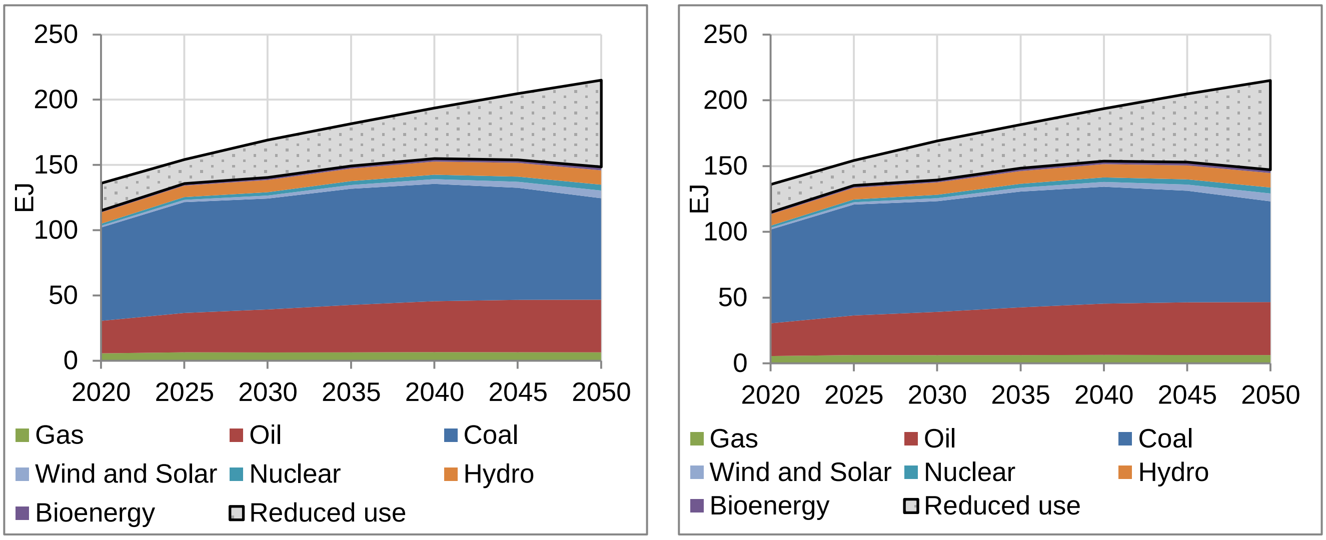
Figure 42 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for Brazil



(a) 2030 (b) 2050

If compared to sectoral trading with other potential partners, however, the magnitude of these technological improvements in Brazil is very limited. For China, the role of low carbon technologies remains unchanged in the presence of a bilateral ETS agreement with Brazil relative to an isolated Chinese sectoral ETS. However, the Chinese sectoral ETS reduces 6.3EJ of energy consumption from 2020 to 2050 compared to the Bra-CHN-Trade scenario. This is depicted in Figure 43. Specifically, it shows a decrease in the consumption of coal by 1% and 2% in 2030 and 2050 respectively. Such a result is in accordance to previous studies (Xu *et al.*, 2017) which indicated that ETS linkage can accelerate the reduction of coal consumption in China, although the effect is smaller in the Brazil-China case.

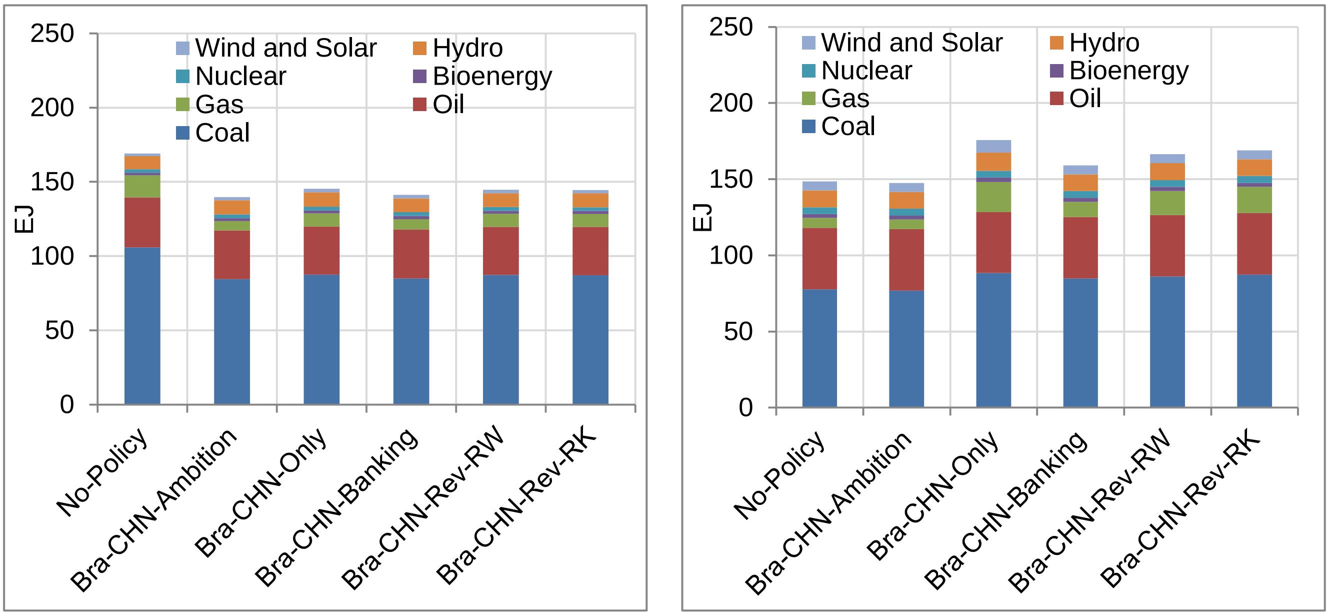
Figure 43 – Primary energy use in China: Domestic ETS vs. Linkage (Bra-CHN-Trade)



(a) CHN-Trade

(b) Bra-CHN-Trade

Figure 44 – Primary Energy Use in 2030 and 2050 under different ETS design scenarios for China



(a) 2030

(b) 2050

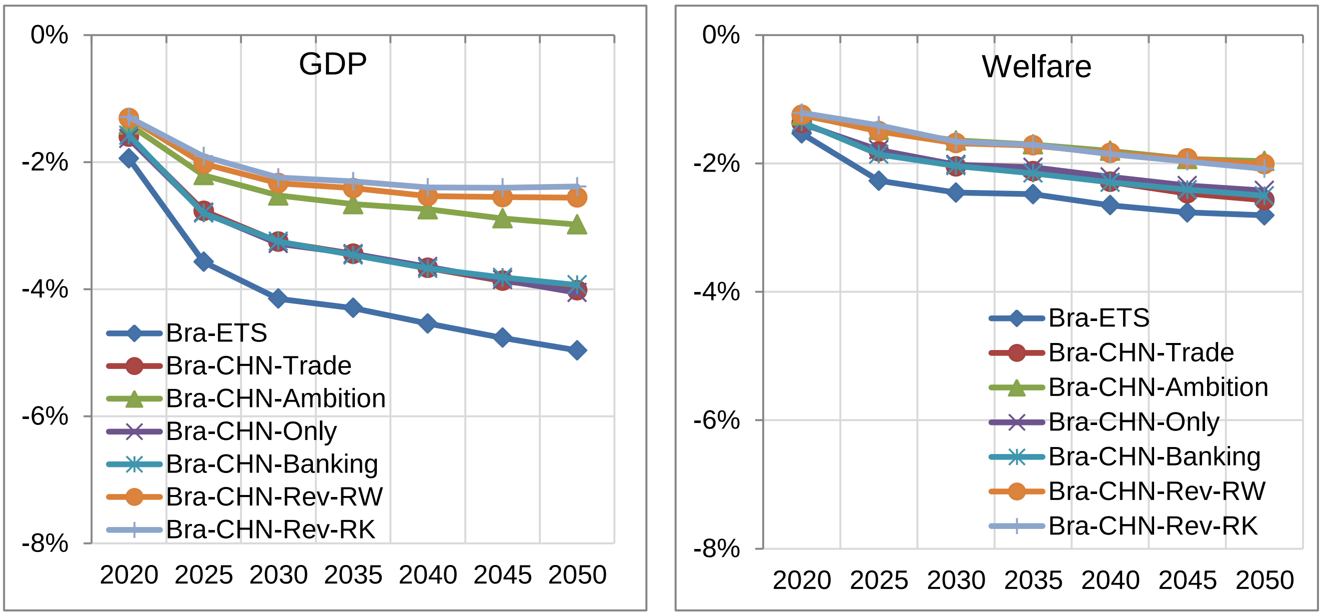
Figure 44 also suggests that among simulated options for the linked ETS system, adopting targets less restricted in Brazil is the best strategy to drive changes in fossil fuel-based energy sources towards renewables in China. This occurs because it pressures ETS sectors in China to seek for technological improvements and cuts fossil fuel use to promote further emissions abatement in order to sell permits to Brazil. Note that the use of alternatives is the same in all linking scenarios, except in Bra-CHN-Only, since it involves high costs to China.

From the economic perspective, linking two heterogeneous developing countries has the advantage to attenuate welfare and GDP implications for the region with the higher level of ambition, Brazil in this case. This is depicted from Figure 45. A sensitivity analysis of GDP growth rate and energy efficiency is provided for the Brazil-China linkage in Appendix E.

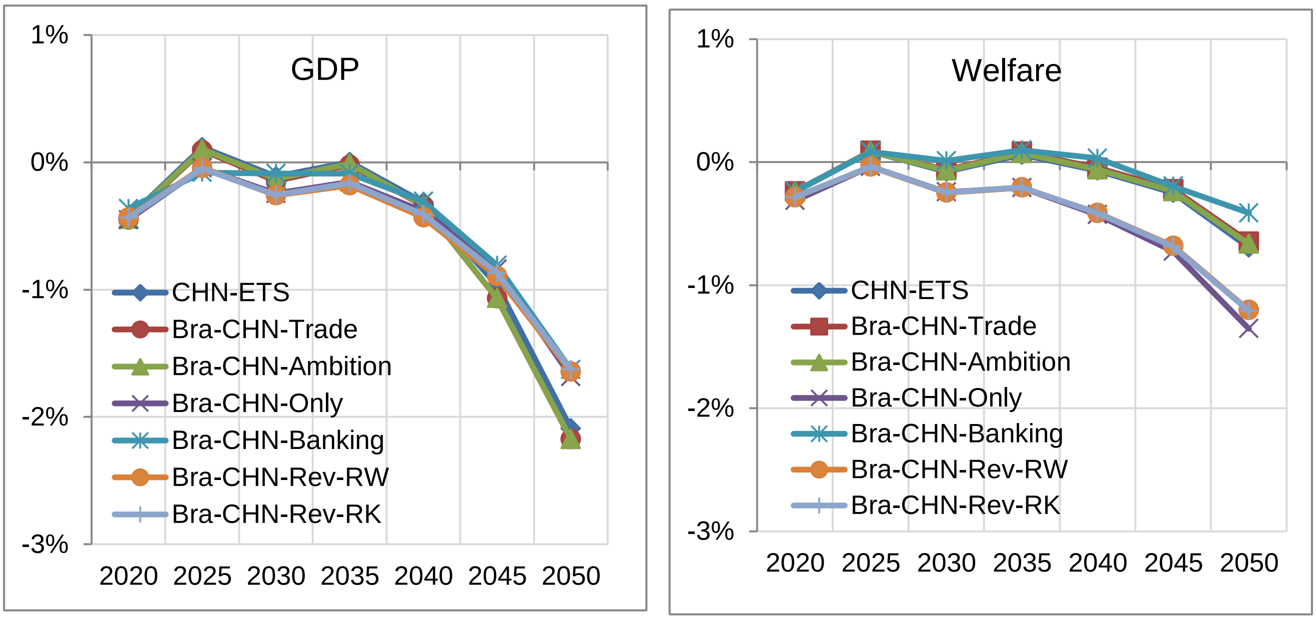
Figure 45a shows that the ETS design matters more for GDP than for welfare since the difference from the scenario with the lowest loss and that with the highest is 0.9% for welfare and approximately 2% for GDP over the period. A climate policy inevitably will raise the overall costs to the economy, yet the lesser the losses, the more opportunities for domestic economic development, which is fundamental for Brazil. If cooperation is agreed in a way to approximate mitigation obligations with China (Bra-CHN-Ambition), Brazil has lower economic impacts. In fact, a sectoral ETS on China under the modelled framework implies minimum effects on the economy.

Another alternative is to allow revenues to be recycled into the production of renewables in Brazil, which is GDP and welfare improving, that is, GDP reduces 2.3% and 2.6% in 2030 and 2050, and welfare 1.7% and 2% compared to No-Policy scenario, respectively. Yet, this mechanism makes the welfare 0.5% lower in China than other scenarios. It is worth noting that the Bra-CHN-Rev-RW also promotes the deepest mitigation of the linked scenarios, through the energy substitution and technological improvements induced by the policy.

Figure 45 – Changes in GDP and Welfare in relation to the No-Policy scenario in (a) Brazil and (b) China under different scenarios



(a) Brazil



(b) China

In summary, evidences suggest that sectoral ETS linking between two developing countries that adopt lower levels of ambition generates limited environmental benefits. Under this circumstance, linking does not contribute with the transition to a low carbon economy. Firstly, this is as a result of the small aggregate reductions achieved compared to linking with a system where mitigation is more aggressive. There are abatement opportunities available in China that makes the linking less costly for Brazil, who decides to maintain the emissions profile and buy permits. For China, the strategy appears to be irrelevant given the volume of emissions it has compared to Brazil's. At the same time, it benefits through the financial transfers received from Brazil. Politically, this scenario could be very attractive for Brazil as long as China does not increase the stringency of the mitigation target. On the other hand, it could signalise the willingness of China to contribute with mitigation in third countries.

This is also valid for Brazil, although it represents the least costly option if a market-based approach is intended. In this case, the most appropriate strategy is to include provisions to allow revenues to be reallocated to the production of renewables in Brazil. Also, this scenario facilitates the deployment of alternative technologies compared to other ETS design options for Brazil and additionally, requires fewer permits to be obtained in the integrated carbon market. Since China presents an export-led profile and is barely affected by the link, the use of revenue recycling as a compensatory measure could minimise distributional effects on Brazil. Nevertheless, the policy framework modelled suggests the need for more stringent mitigation commitments by China, regardless of being involved in a linked ETS system or not.

The results of such a linkage are directly associated to the admitted level of ambition for Brazil and trading partners. As previously mentioned in Chapter 3, to date the Brazilian NDC has not considered economic instruments to help achieve the proposed targets, and has put little emphasis on requirements for energy-intensive industries. For that, there is the Industrial Plan, which imposes a lower mitigation target. In order to approximate estimations to the existing plan, Appendix C includes an analysis of the Brazil-Europe, Brazil-Latin

America and Brazil-China sectoral linkage where the Brazilian ETS is capped by the Industrial Plan target.

For comparability purposes, a global sectoral ETS is further modelled in Appendix D in order to briefly analyse the potential of a global market for participants. Despite being a very unlikely scenario at least in the short term, simulating a worldwide sectoral ETS provides some understanding of the potential effectiveness of global international cooperation to address climate change using market mechanisms, compared to bilateral or multilateral cooperation agreements.

5.4 To link or not: lessons and policy implications

Literature on climate policies highlights that the most significant environmental and economic outcomes of a cap-and-trade scheme are achieved via linkages with other systems. In order to ensure a price signal that incentivises the decarbonisation of the economy, the bilateral cooperation has to be designed appropriately. Both the ETS and the linkage design are context-specific and result from the combination and interplay of political, economic and environmental circumstances and priorities. Hence, negotiating is always challenging.

In the case of the domestic ETS design, some elements are disregarded in the simulations, such as offset, cost-management provisions and differentiation with regard to allocation methods for simplification reasons. For the linkage, the same emissions and sectoral coverage are applied while the cap setting differs among participants. This is a critical element since it determines the price in the trading market. Results indicate that where linkage occurs from the more to the less ambitious trading candidate, it engenders cheaper opportunities to Brazil to purchase permits, thereby reducing abatement costs compared to domestic ETS mitigation.

Hence, the relative level of emissions reductions required in the combined ETS is lower to Brazil than without linking. In fact, capping the ETS based on the Brazilian commitments in the Paris Agreement is as stringent as Europe’s, slightly more stringent than Latin America’s and far above the

Chinese ambition. The policy responses are in accordance with these differences in the carbon constraint imposed in each jurisdiction. This is evidenced in scenarios where Brazil adopts a less strict emission reduction target and overall cost to society diminishes, since mitigation is cheaper.

In the short term, Brazil becomes a buyer of emissions not only if linked to the EU ETS but also if linked to other developing regions, indicating that the intra and inter-trading pattern of ETS linkage is determined by the ETS design, with focus on the level of mitigation. The findings demonstrate a reallocation of emissions and financial transfers among sectors and participants of the linked system, i.e., emissions reductions and carbon permits to a larger extent are transferred from the potential partners to Brazil. As the cap gets gradually stricter in the long-term, Brazil assumes an export-led profile in the link with Europe, as well as in the Brazil-Latin America linkage with banking provisions. Under these circumstances, the marginal cost is below the price and there is stimulus to further reduce emissions, which creates permits to be traded. When the supply of allowances is lower, whether due to the cap being more stringent or the banking-effect, the prices are higher and the signal towards mitigation is stronger.

As pointed out by Gavard, Winchester and Paltsev (2016), unlimited sectoral trading with a developing country ETS, improves the situation for developed regions or at least has limited impact on them, due to their more stringent cap. Similarly, linking with developing countries that diverge in ambition also prevents adverse economic effects from taking place, as in the Brazil-China case. Due to the small volume of emissions regulated in the Brazilian ETS, sectoral trading has little impact on the EU ETS and CHN ETS, and modest implications on emissions of Latin America. Comparatively, linking with China has the larger impact on the equalised price for Brazil in light of the greater differences in abatement cost among them. In this context, results corroborate with the theoretical literature on the cost-effectiveness of the cooperation as well as competitiveness effects.

From the evidence presented, some insights for policy implementation emerge. Firstly, international cooperation is relevant for mitigation but matching with partners whose environmental objectives are similar is fundamental if the main goal is to induce a low carbon economy. In the developed-developing country linkage, this means negotiating a less ambitious target for the developing country whilst introducing mechanisms for revenue recycling to alleviate distributional impacts on the impacted jurisdiction.

As developing countries are starting to engage in linkages, it is important to recognise that linking among developing regions needs to be tailored to the jurisdiction context so to support conditions for decarbonisation and economic development, which prompt a broad public acceptance and confidence in the functioning of the system. Considering the urgent need to reduce emissions, it is suggested to increase the Chinese level of ambition, since it represents a large share of total global emissions and has opportunities for abatement available, particularly because the energy mix is mostly carbon-intensive. In this case, linking with other developing countries such as Brazil would lead to greater technological substitution towards renewables whilst effectively helping to reduce emissions by 2050. To accomplish this objective, the best ETS design is the one that redistributes revenues, in particular as a subsidy to the production of renewables, which also holds for the Brazil-Latin America sectoral linkage.

The Brazilian case is used to illustrate the effects for a developing country to design a sectoral ETS and link it to other systems, whether trading with developed or another developing region. Evidence from this study suggests that for Brazil, linking can lead to significant economic benefits. In addition, it highlights in which partnership the implementation of a linkage is more appropriate to meet mitigation goals at minimum cost, whilst encouraging a broader participation.

Table 20 below summarises the advantages and disadvantages of linking and characterises each proposed trading partnership based on existing requisites in the literature for a successful linkage. Among environmental, economic and political aspects, the modelling results indicate that it is very challenging to address all conditions at once, especially because opportunities for abatement vary across countries for several reasons. The level of development is particularly important, not only to understand if sectoral ETS linkage is appropriate to national circumstances, but also to indicate to what extent the balance between costs and benefits from mitigation via market-based mechanisms enables human development. However, the major determinant on the implications of linking is the level of ambition, which translates to both the energy profile and the level of development of jurisdictions.

Even though energy profile and level of development differ among Brazil and Europe, this linkage is more environmentally effective due to higher overall abatement levels promised along with the fossil-fuel substitution effect it triggers. By linking with Europe, Brazil would benefit from the technical know-how, and signal that developing countries are also willing to tackle climate change through cooperation. At the same time, this linkage could lead to lower political autonomy as some basic rules intrinsic to the EU ETS would end up being transmitted to the Brazilian ETS, for example, in the EU ETS system there is the New Entrant Reserve (NER300) or specific allocation method mechanisms. Moreover, there are additional distributional implications to Brazil.

Considering the ETS design modelled, allowing some flexibility for the developing country appears to be reasonable so as to address the challenges and difficulties of this proposal. Examples could include, accepting less rigid commitments or transferring payments from the allocation of permits to developing countries. Yet, Europe may not perceive the prospects of linking with Brazil as attractive, as opposed to linking to a larger system, such as the Chinese ETS.

Table 20 – Characteristics of each proposed linkage from the modelling exercise and according to the literature

Linkage characteristics	Brazil-Europe	Brazil-Latin America	Brazil-China
Comparable level of ambition	X		
Similar energy profile		X	
Similar level of development		X	
Similar marginal costs of abatement translated into carbon prices	X		
Improved market liquidity due to the broader participation of countries	X	X	
Lower costs to society			X
Incentives to develop low-carbon technologies	X	X	
Greater environmental performance: aggregate emissions reductions	X		
Geographic proximity		X	
Symbol of multilateral efforts to mitigate emissions	X	X	X
Potential for addressing competitiveness issues		X	X
Economic history of cooperation	X	X	X
Politically more acceptable at domestic level			X
Potential for reduced regulatory sovereignty	X		
Technical know-how	X		X
Adverse distributional effects	X	X	X
Risk to endorse reductions targets less consistent with the socially efficient in a global perspective			X

Similarly, the Brazil-China ETS is considerably heterogeneous in terms of energy profile and economic performance but also with regard to counterfactual prices. The most appealing aspect of this linkage is the lower costs for society it promotes, thereby making the cooperation more politically acceptable at domestic level. In fact, there is a long history of economic relations between Brazil and China, especially via the international trade of goods and services that could facilitate negotiation. In light of the recent launch of the Chinese ETS, which tends to provide China with technical expertise, linking of ETS systems has the potential to address competitiveness issues, whilst encouraging other developing countries to take action. As illustrated by the analysis, this is the case if Brazil aims to reduce emissions from electricity and energy-intensive sectors to comply with the stringent targets it has, whereas China (and India) adopt less rigid commitments.

The political argument favours China, as it may prefer to link rather than accept to increase the level of ambition over time. However, the policy may reduce the overall economic costs mainly because China proposes an intensity-based emissions cap, which for the purpose of the model used in this analysis, was translated into an absolute cap which may result in it not being comparatively ambitious. In a linked framework, differences in the type of cap are technically difficult to regulate. Most importantly, contrasting mitigation goals may hinder the aggregate emissions reduction.

Latin America would appear to be the trading partner most closely aligned to Brazil from an energy and economic perspective. As for the link, mitigation targets are less aggressive than those taken on by Brazil. This sectoral ETS linkage among developing countries demonstrates that it is possible to curb emissions at lower costs when the level of mitigation differs across systems. In this case, rather than transferring revenues to a developed country, the distributional effect of linking is a financial flow from Brazil to Latin America that comprises a wider abatement effort within a shared carbon price. The advantage of this regional cooperation is to culminate in a broad participation where competitiveness concerns may be addressed, especially because all sectors face a constraint on emissions, along with transformations of the energy system.

At the same time, this could be a complex negotiation since Latin America has not signalised a willingness to discuss a common climate policy. This approach is indeed very politically challenging in all linked situations. Although the link can be geographically strategic, political support or administrative costs may vary among the heterogeneous group of countries, thereby highlighting potential difficulties for the initiation stage.

This is also applied to the suggested harmonisation of ETS features, as results reveal the challenge of imposing a deep carbon constraint on a small market already very low in emissions. Although upfront alignment of coverage facilitates negotiations of the linkage, perhaps it does not take into consideration the greatest abatement opportunity at the domestic level. Accordingly, it indicates that Brazil may not be seizing other possible mitigation opportunities, such as reducing deforestation and other GHGs, which may be cheaper than curbing emissions only in production sectors, and only based on market instruments.

This fact does not exclude or reduce the need to adopt measures in the electricity and energy-intensive sectors. Once the sectoral ETS progressively mitigates emissions, the focus will shift increasingly towards reducing emissions from other sectors. Therefore, strengthening the climate package with domestic carbon taxes to curb emissions outside the ETS is rather necessary, as are regulatory and technology policies to enhance innovation, or to compensate those sectors disproportionately affected. These additional factors are relevant for further analysis.

CHAPTER 6
CONCLUSIONS AND SUGGESTIONS FOR FUTURE RESEARCH

International cooperation through carbon pricing has become an important framework to address climate change, as highlighted in the Paris Agreement. In light of that, both developed and developing regions are encouraged to adopt market measures in the future, with flexibility to determine the role carbon prices might play in the policy mix. With the number of ETS systems increasing around the world, the question of whether these schemes should be linked, or under what conditions developing countries should take part, is relevant.

To date, experience shows that an ETS comprehensive enough to apply a given emissions constraint on all economic sectors is technically unfeasible. Since Brazil is discussing the implementation of carbon pricing mechanisms, the objective of this thesis is to investigate the impacts of adopting an ETS strategy to reduce emissions and the use of fossil-fuel energy, as well as quantifying the related costs. This is particularly framed in the context of ETS linkage agreements.

We made assumptions on the ETS design features in line with the EU ETS characteristics, as it is the most consolidated system. Thus, to comply with the environmental objectives we applied a sectoral ETS, regulating electricity and energy-intensive sectors, along with a supplementary policy on non-ETS sectors, so as to mimic abatement in those sectors and to prevent carbon leakage. To avoid competitiveness issues associated with carbon leakage towards other regions, we adopted the same hybrid architecture in other jurisdictions of the model. The only exception is the scenario where no ETS is applied to other regions.

This research serves as a basis to evaluate ETS policy proposals between consolidated systems in developed regions, and emerging sectoral ETS systems from developing countries. We consider Europe, Latin America and China as candidates to link with Brazil due to geographic proximity, and/or historical economic relations. Europe has envisaged linking to non-EU emerging trading systems in the future to strengthen the EU ETS. China has just launched the national Chinese ETS (December 2017) and Latin America remains a hypothetical ETS system, despite the growing interest in carbon pricing in the region. Whilst Europe and China have technical knowledge to manage an ETS, Latin America has not initiated any discussion in this regard.

Using a dynamic-recursive computable general equilibrium model, the EPPA6, this policy research has quantitatively evaluated the main economic and environmental implications of a two-way ETS where Brazil is involved. Simulations include an autarky scenario for Brazil and each trading partner, in addition to six other linkage scenarios (Bra-Trade, Bra-Ambition, Bra-Only, Bra-Banking, Bra-Rev-RW, Bra-Rev-RK). Additionally, a reference scenario is modelled (No-Policy) where no mitigation policy is implemented.

The modelling results demonstrated that differences in carbon prices are eliminated through the link and, corroborating with theoretical literature, Brazil benefits from a lower carbon price if it links to any partner. In the Brazil-Europe case, carbon prices equalise at approximately US\$140/tCO_2 in 2030 and US\$850/tCO_2 in 2050. The highest price among linked scenarios in 2030 is US\$173/tCO_2, corresponding to the ETS design where allowances can be banked and carried forward over periods. Banking provisions initially increase the costs of carbon due to the anticipation of abatement; however, it induces a carbon price of US\$380/tCO_2 in 2050. The costs for meeting the climate obligations through emissions trading are still very high in other scenarios, at US\$840/tCO_2 on average by 2050. In this case, Europe benefits from linking as trading allows committing to more stringent targets whilst sharing a carbon price 55% lower than in the isolated EU ETS.

Europe is the most displays heterogeneity in relation to energy and macroeconomic patterns compared to Brazil. Although emissions reductions are facilitated in this linkage, it implies significant effects in terms of GDP and welfare, particularly for Europe which faces approximately a 10% loss in the more costly scenario (Bra-EU-Trade) in 2050. One alternative is to include revenue recycling in the ETS design, since it alleviates adverse economic effects in Europe, which is 0.4% and 6.9% of GDP loss, and 1.3% and 6.3% of welfare loss in 2030 and 2050, respectively.

Such significant costs result from the stringent level of targets adopted in both jurisdictions, and the availability of opportunities for abatement as the Brazilian ETS sectors are significantly more decarbonised than Europe's. Results from the Bra-EU-Ambition scenario give a clear example of the high level of effort required for Brazil. If the link is negotiated to accept lower mitigation targets, being in line with the principle of common but differentiated responsibilities, Brazilian GDP declines 3.2% and 3.7% in 2030 and 2050 respectively, whereas welfare decreases by 1.9% in 2030 but grows 0.4% in 2050. Similar effects occur in the case of recycling revenue to the production of renewables in Brazil, where there is a smaller reduction in both GDP and welfare levels, 2.9% and 2% in 2030, and 2.7% and 0.5% in 2050, respectively.

In this investigation, the difference in stringency of ETS targets between a developed and developing region implies different trading patterns over the period. From 2020-2030 Brazil assumes an importer-oriented profile, with payments for allowances accruing to Europe of approximately US$3 billion. Thereafter, Brazil becomes a net exporter of allowances, which is aligned to the literature, i.e. developing countries pursuing a permit exporter pattern. Under this perspective, transfers to Europe total more than US$60 billion. To a certain extent, this financial flow compensates for the early costs incurred during the linkage. Ultimately, linking triggers a fossil-fuel substitution effect in both countries, which is essential for the decarbonisation of the economy.

A sectoral link with developing regions (China and Latin America) has the advantage of diminishing the carbon costs for the Brazilian ETS. In both linkages, Brazil continues to have an allowances importer profile, with financial transfers to Latin America and China totalising on average US$8 and US$4.5 billion in 2050, respectively. Whilst in the Brazil-Latin America linkage scenario carbon prices decrease 56% from the autarky scenario in 2030, the Brazil-China linkage carbon price is reduced by 85% in the same period.

A linkage involving developing countries improves the economic performance of Brazil, particularly resulting from the less aggregated emissions reductions required. This is because the Paris Agreement commitments of Latin America and China are less ambitious than those for Brazil and the EU. This result indicates that the climate cooperation between developing countries, and considering their lower national targets, is less costly than linking to a developed ETS. In both developing-developing region ETS linkages, a trade deal that recycles payments of permits to renewable energy production in Brazil would be economically advantageous for Brazil but also for Latin America. For China, accepting to transfer those revenues to Brazil is GDP improving in the long-run. Otherwise, economic benefits arise from Brazil adopting a lower mitigation target in the Brazil-China linkage.

In the context of the ETS scenarios modelled, it appears that economic benefits would be an aspect which could increase interest for Brazil to link, particularly if there are associated gains of trade. Among the proposed trading partners, the Brazil-China sectoral link fits this prerequisite and tends to be more politically acceptable. When linking with China, concerns about conditions of international competitiveness are partially addressed, contributing to further acceptance. Brazil would prefer to negotiate the climate agreement and hence, to share the burden of the policy, as long as China commits to less stringent targets. From the Chinese perspective, the different levels of ambition could be easily accepted on the basis of gains from selling permits. More importantly, a joint commitment would send a strong political signal on the willingness to contribute with mitigation in third countries by setting an unambitious cap. Ultimately, it could become a first step towards closer cooperation with other large emitters in the developing world, such as India.

However, resolving the problem of global climate change requires China to engage with deeper mitigation. In the Brazil-China linkage, the political acceptance would dominate, rather than the environmental protection, as it implies a lower abatement level. On the other hand, if in the short run Brazil prioritises a climate strategy committed to deep decarbonisation based on the development of low-carbon technologies, linking to the EU ETS is the most appropriate choice since environmental integrity is also a priority for Europe. As for Latin America, political resistance and technical limitations for using market mechanisms to address climate issues makes the proposed linkage extremely unlikely before 2030.

Therefore, the analysis in this thesis suggests that the level of ambition is crucial to determine the benefits of linking for a developing country, being a precondition for deciding the most appropriate partner to link with. Evidence indicates more political, than economic and environmental advantages for the considered trading partners.

Moreover, this thesis contributes to the scientific literature by showing the appropriateness of sectoral ETS linkage from the developing country perspective. If Brazil decides to cap emissions of electricity and energy-intensive sectors via an ETS at a highly ambitious level, as committed to in the Paris Agreement, there are benefits in linking. An option would be to participate in a global ETS, which reduces the costs of the climate policy. If linked to a developed country ETS, an inflow of revenues is envisaged from selling revenues in the long term, and it is possible to curb emissions whilst changing energy use patterns towards less carbon-intensive technologies. On the other hand, joining developing countries, in which the carbon constraint is less restrictive, is economically more efficient as it reduces the adverse impacts on the economy, but the magnitude varies according to the ETS design implemented.

The study has some recommendations for policy-making. Indeed, a common carbon price has proven to provide flexibility to Brazil and may configure an important element of the climate policy package aimed to foster a transition to a low-carbon economy. In the illustrative case of Brazil, the alignment of sectoral coverage may not seize the main domestic mitigation opportunities, considering that the share of low-carbon technologies in the energy mix is currently higher than fossil fuel-based sources in Brazil. This, in turn, makes the electricity and energy intensive sectors more decarbonised than, for example, those of the EU or China. In this instance, the link could envisage the incorporation of other GHGs and additional sectors, notably deforestation and land use change in the case of Brazil, where cheaper mitigation opportunities are available. This could be introduced in the system via offsets.

Once the sectoral ETS progressively mitigates emissions, the focus will shift increasingly towards reducing emissions from other sectors. Thereupon, strengthening the climate package with domestic carbon taxes to curb emissions outside the ETS is rather necessary, as are regulatory and technology policies to enhance innovation, or to compensate those sectors disproportionately affected. Modelling the ETS and related linkages without the participation of other regions could be interesting, so as to investigate carbon leakage effects. For a further stage of the research, recalibrating the EPPA6 model to represent the actual historical projection of emissions (hindcast), is envisioned. It was not considered for this thesis but is relevant for further research in this area.

Another recommendation is to allow a level of abatement more appropriate to mitigation opportunities available in the developing country, if linked to a developed country ETS. As scenarios with lower ambition show, including simulations based on the Industrial Plan targets in Appendix C, there is an increase in the economic performance of Brazil. Most importantly, it is desirable to introduce a degree of flexibility to increase acceptability while reducing distributional impacts at local level, such as revenue recycling. In future policy iterations, extending the use of revenue for both participants, could provide a more realistic approach.

To address the global climate problem and promote greater abatement, a sectoral ETS linkage with only developing countries (Brazil, Latin America and China), designed specifically in relation to their energy and economic profiles, may be relevant. Since the incentives for linking at different levels are likely to continue post-2020, future research could also incorporate other worldwide emitters to form a multilateral agreement (or a Climate Club), including the participation of Brazil through indirect linkage.

Despite the potential positive effects to Brazil of linking, there appears to be some challenges for its implementation, specifically in terms of regulatory convergence. As discussed, to a certain extent the linkage framework involves a loss of regulatory control because adjustments in the permit price, or changes in design features, propagate from one scheme to the other, which could compromise national policy objectives. At the governance level, a high importance needs to be placed on regulatory coordination, in order to develop the necessary technical, legal and institutional capabilities to support enforcement of the climate policy. These aspects are conceivable for future investigations.

The approach modelled does not consider the costs or benefits associated with avoiding climate change, climate adaptation, or for other policies to support technological change at the intra-industry level. However, it does configure a first approximation on how developing countries could design their ETS, and incorporate carbon pricing and trading arrangements, with the aim of reducing emissions in as cost effective a fashion as possible.

REFERENCES

Acworth, W. *et al.* (2017) 'Emissions trading and the role of a long-run carbon price signal: Achieving cost-effective emission reductions under an emissions trading system', (June), pp. 2–45. Available at: https://icapcarbonaction.com/en/?option=com_attach&task=download&id=491.

Aguiar, A., Narayanan, B. and McDougall, R. (2016) 'An Overview of the GTAP 9 Data Base', *Journal of Global Economic Analysis*.

Ahlberg, M. *et al.* (2013) *Linking Different Emissions Trading Systems-Current State and Future Perspectives*. Available at: www.dehst.de/EN.

Aldy, J. E. and Stavins, R. N. (2011) 'Using the Market to Address Climate Change: Insights from Theory and Experience', *National Bureau of Economic Research Working Paper Series*, No. 17488. Available at: http://www.nber.org/papers/w17488.

Aldy, J. and Pizer, W. . (2009) 'The Competitiveness Impacts of Climate Change Mitigation Policies', *Pew Center on Global Climate Change*, pp. 1–41.

Almeida, L. T. (1997) 'O debate internacional sobre instrumentos de política ambiental e questões para o Brasil', in. São Paulo: 2º Encontro da Sociedade Brasileira de Economia Ecológica (Eco-Eco), pp. 3–21.

Ambec, S. *et al.* (2013) 'The Porter Hypothesis at 20: Can Environmental Regulation Enhance Innovation and Competitiveness?', *Review of Environmental Economics and Policy*, 7(1), pp. 2–22.

Andersen, K. S. and Termansen, L. B. (2013) *Bottom-up and top-down modelling approach*, *IntEract model*. 4. Available at: http://www.ens.dk/sites/ens.dk/files/info/facts-figures/scenarios-analyses-models/models/IntERACT/wp04_-_bottom-up_and_top-down_modelling_approach.pdf.

Andersen, M. S. and Ekins, P. (eds) (2009) *Carbon-Energy Taxation: Lessons From Europe*. New York: Oxford University Press.

Anderson, B. and Di Maria, C. (2011) 'Abatement and Allocation in the Pilot Phase of the EU ETS', *Environmental and Resource Economics*, 48(1), pp. 83–103.

Anderson, R. (2002) 'Incentive-based policies for environmental management in developing countries', *Resources for the Future Issue Brief*, pp. 02–07. Available at: http://206.205.47.99/RFF/Documents/RFF-IB-02-07.pdf.

Anger, N. (2008) 'Emissions trading beyond Europe: Linking schemes in a post-Kyoto world', *Energy Economics*.

Anger, N. and Sathaye, J. (2008) *Reducing Deforestation and Trading Emissions: Economic Implications for the Post-Kyoto Carbon Market*, *Ssrn*. 8–16.

Anthoff, D. and Tol, R. S. J. (2013) 'The uncertainty about the social cost of carbon: A decomposition analysis using fund', *Climatic Change*.

Armington, P. (1969) 'A Theory of Demand for Products Distinguished by Place of Production.', *Staff Papers (International Monetary Fund)*, 16(1), pp. 159–178.

Arrow, K. J. and Debreu, G. (1954) 'Existence of an Equilibrium for a Competitive Economy', *Econometrica*, 22(3), pp. 265–290.

Averchenkova, A. and Bassi, S. (2016) 'Beyond the targets: assessing the political credibility of pledges for the Paris Agreement', *LSE Policy Brief*, (February), p. 60. Available at: http://www.lse.ac.uk/GranthamInstitute/wp-content/uploads/2016/01/Averchenkova-and-Bassi-2016.pdf.

Babiker, M. H. (2005) 'Climate change policy, market structure, and carbon leakage', *Journal of International Economics*, 65(2), pp. 421–445.

Babiker, M., Reilly, J. and Viguier, L. (2004) 'Is International Emission Trading Always Beneficial?', *The Energy Journal*, 25(2), pp. 33–56. Available at: http://18.7.29.232/handle/1721.1/3628.

Bakker, K. (2010) 'Neoliberalizing Nature? Market Environmentalism in Water Supply in England and Wales', *Annals of the Association of American Geographers*, 95(3), pp. 542–565.

Balistreri, E. J. *et al.* (2015) *Carbon policy and the structure of global trade*. 383. Oldenburg.

Baranzini, A. *et al.* (2017) 'Carbon pricing in climate policy: seven reasons, complementary instruments, and political economy considerations', *Wiley Interdisciplinary Reviews: Climate Change*, 8(4).

Barata, P. and Kachi, A. (2016) *Study on approaches to incorporation of mitigation contributions in international market mechanisms, including through development standards for setting emissions reference levels: International emission transfers after 2020*.

Baron, R. and Bygrave, S. (2002) *Towards International emissions trading: design implications for linkages*. Available at: http://www.oecd.org/env/cc/.

Barrett, S., Carraro, C. and Melo, J. de (2015) 'Introduction', in Barrett, S., Carraro, C., and Melo, J. de (eds) *Towards a Workable and Effective Climate Regime*. CEPR Press and Fondation pour les études et recherches sur le développement international (Ferdi).

Barrett, S. and Moreno-Cruz, J. B. (2015) 'The alternatives to unconstrained climate change: Emission reductions versus carbon and solar geoengineering', in Barrett, S., Carraro, C., and Melo, J. de (eds) *Towards a Workable and Effective Climate Regime*. London: Centre for Economic Policy Research, pp. 353–365.

Bassi, S. *et al.* (2017) 'Credible, effective and publicly acceptable policies to decarbonise the European Union (Final report)'. Available at: http://www.lse.ac.uk/GranthamInstitute/publication/credible-effective-publicly-acceptable-policies-decarbonise-european-union-final-report/.

Van Beers, C. and Van den Bergh, J. (1997) 'An empirical multi- country analysis of the impact of environmental regulations on foreign trade flows', *Kyklos*, 50(1), pp. 29–46.

Belini, L. (2005) *Mudanças Climáticas: Politicas e Negociações Internacionais*. Universidade Estadual Paulista, Marilia.

BEN (2015) *Balanço energético nacional*. Brasília.

Beuermann, C. *et al.* (2017) *Considering the Effects of Linking Emissions Trading Schemes: A Manual on Bilateral Linking of ETS*. Available at: https://www.dehst.de/SharedDocs/downloads/EN/emissions-trading/Linking_manual.pdf?__blob=publicationFile&v=3.

Bodansky, D. M. *et al.* (2014) *Facilitating Linkage of Heterogeneous Regional, National, and Sub-National Climate Policies Through a Future International Agreement Faculty Research Working Paper Series*. Available at: www.hks.harvard.edu.

Boden, T. ., Marland, G. and Andres, R. . (2010) *Global, Regional, and National Fossil-Fuel CO2 Emissions*. USA.

Boff, L. (2015) 'A enganosa proposta da COP-21', *Revista ECO21*, 229. Available at: http://www.eco21.com.br/textos/textos.asp?ID=3755.

Böhm, S., Misoczky, M. C. and Moog, S. (2012) 'Greening Capitalism? A Marxist Critique of Carbon Markets', pp. 1–22.

Böhringer, C. and Löschel, A. (2003) *Climate Policy Beyond Kyoto: Quo Vadis? A Computable General Equilibrium Analysis Based on Expert Judgements*. 3–9.

Böhringer, C. and Rutherford, T. F. (2005) 'Integrating Bottom-Up into Top-Down: A Mixed Complementarity Approach', *Ssrn*, (05).

Bollen, J. *et al.* (2009) 'Local air pollution and global climate change: A combined cost-benefit analysis', *Resource and Energy Economics*, 31(3), pp. 161–181. Available at: https://ac.els-cdn.com/S092876550900013X/1-s2.0-S092876550900013X-main.pdf?_tid=b34fc1ab-8c82-437d-a7c9-57dfc7f77d8c&acdnat=1532546172_60c38876afbc6bfd1f3474eaad7a35cb.

Bosetti, V. *et al.* (2013) 'Incentives and stability of international climate coalitions: An integrated assessment', *Energy Policy*.

Bowen, A. (2015) *Carbon pricing : how best to use the revenue ?*

Bowen, A., Campiglio, E. and Tavoni, M. (2014) 'A macroeconomic perspective on climate change mitigation: meeting the financial challenge', *Climate Change Economics*.

Branger, F. *et al.* (2014) *EU ETS, free allocations and activity level thresholds, the devil lies in the details*. 169. Available at: http://www.lse.ac.uk/grantham.

Bresnihan, P. (2017) 'Valuing Nature–Perspectives and Issues', (11).

Brooke, A. *et al.* (1998) *GAMS A User Guide*. Washingto, DC.

Bureau, B. (2011) 'Distributional effects of a carbon tax on car fuels in France', *Energy Economics*, 33(1), pp. 121–130.

Burtraw, D. *et al.* (2013) *Linking by Degrees: Incremental Alignment of Cap-and-Trade Markets*. Available at: www.rff.org.

Calel, R. and Dechezleprêtre, A. (2012) 'Environmental Policy and Directed Technological Change: Evidence from the European carbon market', *Centre for Climate Change Economics and Policy Working Paper*, (87).

Calzadilla, A., Rehdanz, K. and Tol, R. S. J. (2011) 'Trade Liberalisation and Climate Change : A CGE Analysis of the Impacts on Global Agriculture', (381), p. 24. Available at: http://www.esri.ie/UserFiles/publications/WP381/WP381.pdf.

Caney, S. and Hepburn, C. (2011) 'Carbon Trading: Unethical, Unjust and Ineffective?', *Royal Institute of Philosophy Supplement*, 69(59), pp. 201–234.

Carbone, J. C., Helm, C. and Rutherford, T. F. (2009) 'The case for international emission trade in the absence of cooperative climate policy', *Journal of Environmental Economics and Management*.

Carraro, C. and Favero, A. (2009) 'The economic and financial determinants of carbon prices', *Finance a Uver - Czech Journal of Economics and Finance*, 59(5), pp. 396–409.

Carvalho, M. M. de *et al.* (2018) 'Simulação Dos Impactos Econômicos Da Proposta Brasileira Na Cop21: Uma Abordagem De Equilíbrio Geral Computável', in *Anais do XLIV Encontro Nacional de Economia [Proceedings of the 44th Brazilian Economics Meeting]*. ANPEC (Brazilian Association of Graduate Programs in Economics).

Carver, T., Dawson, P. and Kerr, S. (2017) *Including forestry in an Emissions Trading Scheme: Lessons from New Zealand*. 17–11. New Zealand. Available at: https://motu.nz/assets/Documents/our-work/environment-and-agriculture/climate-change-mitigation/emissions-trading/Forestry-and-the-ETS-Exec-Summary.pdf.

Castro, L. M. (2013) *Análise de mercado de carbono no Brasil: aspectos de eficiência e equidade*. Universidade Federal do Rio de Janeiro.

Chapman, J. (2009) 'Linking a United States Greenhouse Gas Cap-and-trade system and the European Union's emissions trading scheme', *Vermont Journal of Environmental Law*, 11.

Chen, Y.-H. H. *et al.* (2015) *The MIT EPPA6 Model: Economic Growth, Energy Use, and Food Consumption*. Available at: http://globalchange.mit.edu/.

Chevallier, J. (2012) 'Econometric analysis of carbon markets: The European Union emissions trading scheme and The Clean Development Mechanism', *Economics Bulletin*, 31(4), p. 2011.

Choi, Y., Liu, Y. and Lee, H. (2017) 'The economy impacts of Korean ETS with an emphasis on sectoral coverage based on a CGE approach', *Energy Policy*. Elsevier Ltd, 109(December 2016), pp. 835–844.

Clarke, L. *et al.* (2016) 'Long-term abatement potential and current policy trajectories in Latin American countries', *Energy Economics*.

ClimateWire (2013) 'New Australian government dismantling climate policies', September.

Coase, R. H. (1960) 'The Problem of Social Cost', *Journal of Law and Economics*, pp. 1–44.

Cole, M. A. (2004) 'Trade, the pollution haven hypothesis and the environmental Kuznets curve: Examining the linkages', *Ecological Economics*, 48(1), pp. 71–81.

Comendant, C. and Taschini, L. (2014) *Submission to the inquiry by the House of Commons Select Committee on Energy and Climate Change on 'Linking Emissions Trading Systems' Policy paper*. Available at: http://www.cccep.ac.uk.

Commission on the Economy, G. (2014) *Better growth, Better climate:the New Climate Economy Report -The Global Report*. Washingto, DC. Available at: http://newclimateeconomy.report/2014/wp-content/uploads/sites/2/2014/08/NCE-Global-Report_web.pdf.

Coria, J. and Sterner, T. (2008) *Tradable Permits in Developing Countries*. Washingto, DC.

Cossa, P. (2004) *Uncertainty Analysis of the Cost of Climate Policies*. Cambridge, Massachusetts.

Costanza, R. *et al.* (2014) *An Introduction to Ecological Economics*. Second. CRC Press.

CRP (2017) *Handbook on Carbon Pricing Instruments*. Available at: https://www.climaterealityproject.org/sites/climaterealityproject.org/files/HandbookonCarbonFinancing_Final_May16.pdf.

Dechezleprêtre, A., Martin, R. and Bassi, S. (2016) 'Climate change policy, innovation and growth', *Grantham Research Institute on Climate Change and the Environment and Global Green Growth Institute Policy Brief*, (January), p. 30. Available at: http://www.lse.ac.uk/GranthamInstitute/wp-content/uploads/2016/01/Dechezlepretre-et-al-policy-brief-Jan-2016.pdf.

Dechezleprêtre, A. and Sato, M. (2017) 'The impacts of environmental regulations on competitiveness', *Review of Environmental Economics and Policy*.

Dellink, R. *et al.* (2014) 'Towards global carbon pricing Direct and indirect linking of carbon markets'. doi: 10.1787/eco_studies-2013-5k421kk9j3vb.

Demailly, D. and Quirion, P. (2008) 'European Emission Trading Scheme and competitiveness: A case study on the iron and steel industry', *Energy Economics*, 30(4), pp. 2009–2027.

Dimaranan, B. V and Mcdougall, R. A. (2006) 'Guide to the GTAP Data Base', in Dimaranan, B. V (ed.) *Global Trade, Assistance, and Production: The GTAP6 Data Base*. West Lafayette: Center for Global Trade Analysis, Purdue University.

Dimitrov, R. S. (2016) 'The Paris agreement on climate change: Behind closed doors', *Global Environmental Politics*.

Doda, B., Quemin, S. and Taschini, L. (2018) *Linking Permit Markets Multilaterally* *.

Doda, B. and Taschini, L. (2017) ‘Carbon Dating: When Is It Beneficial to Link ETSs?’, *Journal of the Association of Environmental and Resource Economists*.

Doda, B., Taschini, L. and Druce, V. (2017) ‘Should the UK stay or should it go? The consequences of a divorce with the EU ETS’, *News & commentaries*, 14 February. Available at: http://www.lse.ac.uk/GranthamInstitute/news/should-the-uk-stay-or-should-it-go-the-consequences-of-a-divorce-with-the-eu-ets/.

Domingues, E. P., Magalhães, A. . and Carvalho, T. S. (2014) ‘Política industrial e os custos de redução de emissões de Gases de Efeito Estufa’, *Prêmio CNI de Economia*.

Dryzek, J. S., Norgaard, R. B. and Schlosberg, D. (2012) ‘Climate Change and Society: Approaches and Responses’, *The Oxford Handbook of Climate Change and Society*, (April 2018), pp. 1–18.

Duan, M. and Zhou, L. (2017) ‘Key issues in designing China ’ s national carbon emissions trading system’, *Economics of Energy and Environmental Policy*, 6(2).

Dudek, D. and Golub, A. (2003) ‘“Intensity” targets: pathway or roadblock to preventing climate change while enhancing economic growth?’, *Climate Policy*, 3, pp. S21–S28.

Eden, A. *et al.* (2016) *Benefits of Emissions Trading*, *ICAP*. Berlin, Germany.

Edenhofer, O. *et al.* (2009) *The Economics of Decarbonization. Report of the RECIPE project*. Postdam.

Edenhofer, O. *et al.* (2015) *Climate Change 2014 Mitigation of Climate Change Summary for Policymakers Technical Summary Part of the Working Group III Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change Edited by*.

Edenhofer, O., Flachsland, C. and Marschinski, R. (2007) *Towards a global CO 2 market - An economic analysis*. Postdam.

EEA (2016) *Overall progress towards the European Union’s 20-20-20 climate and energy targets. In Trends and projections in Europe 2016 – Tracking progress towards Europe’s climate and energy targets*. Brussels. Available at: https://www.eea.europa.eu/themes/climate/trends-and-projections-in-europe/1-overall-progresstowards-%0Athe.

EEA (2018) *Data and Maps*, *European Environmental Agency*. Available at: https://www.eea.europa.eu/data-and-maps (Accessed: 5 July 2018).

Egenhofer, C. (2007) ‘The Making of the EU Emissions Trading Scheme: Status, Prospects and Implications for Business’, *European Management Journal*, 25(6), pp. 453–463.

Ellerman, A. D., Marcantonini, C. and Zaklan, A. (2016) ‘The european union emissions trading system: Ten years and counting’, *Review of Environmental Economics and Policy*, 10(1), pp. 89–107.

Ellerman, A. D., Valero, V. and Zaklan, A. (2015) *An Analysis of Allowance Banking in the EU ETS*, *Ssrn*.

Ellerman, D. (2009) ‘The EU emission trading scheme: A prototype global system?’, in *Post-Kyoto International Climate Policy: Implementing Architectures for Agreement: Research from the Harvard Project on International Climate Agreements*.

Ellerman, D. and Buchner, B. (2006) *Over-Allocation or Abatement? A Preliminary Analysis of the EU Emissions Trading Scheme Based on the 2006 Emissions Data*. 141.

Ellerman, D., Buchner, B. and Carraro, C. (2007) *Allocation in the European Emissions Trading Scheme: Rights, rents and fairness*. Cambridge.

Ellerman, D., Convery, F. and Perthuis, C. (2010) *Pricing Carbon: The European Union Emissions Trading Scheme*. Cambridge: Cambridge University Press.

Ellerman, D., Jacoby, H. D. and Prinn, R. G. (2009) *MIT Joint Program on the Science and Policy of Global Change The EU’s Emissions Trading Scheme: A Prototype Global System? The EU’s Emissions Trading Scheme: A Prototype Global System?*

Ellerman, D. and Joskow, P. (2008) *The European Union’s Emissions Trading System in Perspective*, *Pew Center on Global Climate Change*. Washingto, DC.

Espagne, E. (2015) ‘Climate Clubs and COP21: Foes or Allies?’, in Sirkis, A. et al. (eds) *Moving the trillions. A debate on positive pricing of mitigation actions*. Rio de Janeiro: Centro Brasil no Clima Ed./CIRED, Nogent-sur-Marne.

EU (2016) *EU Reference Scenario 2016 – Energy, transport and GHG emissions - Trends to 2050*.

European Comission (2009) *Towards a comprehensive climate change agreement in Copenhagen. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions*. Brussels.

European Commision (2015) *European Union Transaction Log - EUCO 169/14*. Brussels. Available at: http://ec.europa.eu/environment/ets/.

European Commission (2011) *Emission Database for Global Atmospheric Research (EDGAR), release version 4.2*. Available at: http://edgar.jrc.ec.europa.eu.

European Commission (2013) ‘Green Paper - A 2030 framework for climate and energy policies’, *COM(2013) 169 final*, pp. 1–16. Available at: http://ec.europa.eu/clima/policies/strategies/2030/documentation_en.htm.

European Commission (2015a) *Decision (EU) 2015/1814 of the European Parliament and of the Council of 6 October 2015 concerning the establishment and operation of a market stability reserve for the Union greenhouse gas emission trading scheme and amending Directive 2003/87/EC*.

European Commission (2015b) *Submission by Latvia and the European*

Commission on behalf of the European Union and its Member States.

European Commission (2018) *Auctioning*. Available at: https://ec.europa.eu/clima/policies/ets/auctioning_en#tab-0-0 (Accessed: 30 August 2018).

Falkner, R. (2016) 'The Paris Agreement and the new logic of international climate politics', *International Affairs*, 92(5), pp. 1107–1125.

Fankhauser, S. and Hepburn, C. (2010) 'Designing carbon markets, Part II: Carbon markets in space', *Energy Policy*.

Fankhauser, S. and McCoy, D. (1995) 'Modelling the economic consequences of environmental policies', in Folmer, H., Gabel, L., and Opschoor, J. (eds) *Principles of Environmental and Resource Economics: A Guide to Decision Makers and Students*. Aldershot, England: Edward Elgar, pp. 253–275.

Farid, M. *et al.* (2016) *After Paris: Fiscal, Macroeconomic, and Financial Implications of Climate Change*.

Feijó, F. T. and Porto Jr., S. (2009) 'Protocolo de Quioto e o Bem Estar Econômico no Brasil Uma Análise Utilizando Equilíbrio Geral Computável', *Análise Econômica (UFRGS)*, 51, pp. 127–154.

Feld, B. and Galiani, S. (2015) 'Climate change in Latin America and the Caribbean: Policy options and research priorities Research at the policy frontier in Latin America: Health, Education, Infrastructure and Housing and Climate Change Sebastian Galiani', *Latin American Economic Review*.

Feng, K. . b *et al.* (2010) 'Distributional effects of climate change taxation: The case of the UK', *Environmental Science and Technology*, 44(10), pp. 3670–3676.

Ferreira Filho, J. B. (2011) 'Introdução aos Modelos Aplicados de Equilíbrio Geral: Conceitos, Teoria e Aplicações', in Cruz, B. de O. et al. (eds) *Economia Regional e Urbana. Teorias e Métodos com ênfase no Brasil*. 1st edn. IPEA, pp. 375–400.

Ferreira Filho, J. B. and Horridge, M. (2017) *Biome composition in deforestation deterrence and GHG emissions in Brazil*. g-274.

Fisher, C. (2003) 'Combining Rate-Based and Cap-and-Trade Emissions Policies', *Climate Policy*, 3(Supplement 2).

Flachsland, C. *et al.* (2008a) *Developing the International Carbon Market. Linking Options for the EU ETS*.

Flachsland, C. *et al.* (2008b) *Developing the International Carbon Market. Linking Options for the EU ETS Report to the Policy Planning Staff in the Federal Foreign Office*.

Flachsland, C., Marschinski, R. and Edenhofer, O. (2009a) 'Global trading versus linking: Architectures for international emissions trading', *Energy Policy*.

Flachsland, C., Marschinski, R. and Edenhofer, O. (2009b) 'To link or not to link: Benifits and disadvantages of linking cap-and-trade systems', *Climate Policy*.

Foley, D., Rezai, A. and Taylor, L. (2013) ‘The social cost of carbon emissions: Seven propositions’, *Economics Letters*, 121(1), pp. 90–97.

França, F. and Gurgel, A. C. (2018) *Impactos econômicos de políticas climáticas europeias e americanas sobre a economia brasileira*.

Freebairn, J. (2010) ‘Carbon taxes vs tradable permits: efficiency and equity effects for a small open economy’, *Tax Reform in Open Economies: International and Country Perspectives*, (February), p. 219.

Furtado, F. (2017) *REDD+ The Carbon Market and California-Acre-Chiapas Cooperation: Legalizing mechanisms of dispossession*.

Gavard, C. *et al.* (2011) ‘What to expect from sectoral trading: a US-China example’, *Climate Change Economics*.

Gavard, C. (2013) *Analyse Economique des Mécanismes Possibles de Couplage du Marché Carbone Européen avec les Pays Emergents*. Available at: https://tel.archives-ouvertes.fr/tel-01057084.

Gavard, C. *et al.* (2013) *Limited Sectoral Trading between the EU ETS and China*.

Gavard, C., Winchester, N. and Paltsev, S. (2016) ‘Limited trading of emissions permits as a climate cooperation mechanism? US–China and EU–China examples’, *Energy Economics*.

Gielen, A. ., Koutstaal, P. . and Vollebergh, H. R. . (2002) ‘Comparing Emissions Trading with Absolute and Relative Targets’, in *2nd CATEP Workshop on the Design and Integration of National Tradable Permit Schemes for Environmental Protection*. London: University College London.

Gilbert, A. *et al.* (2006) *Comparative analysis of national allocation plans for phase I of the EU ETS*. London (UK).

Godfray, H. C. J. *et al.* (2010) ‘Food Security: The Challenge of Feeding 9 Billion People’, *Science*, 327(5967), pp. 812–818.

Goers, S. and Pflüglmayer, B. (2012) ‘Post-Kyoto Global Emissions Trading: Perspectives for Linking National Emissions Trading Schemes with the EU ETS in a Bottom-Up Approach’, 3, pp. 69–79.

Goodstein, E. (2010) *Economics and the environment*.

Görlach, B. (2013) *What constitutes an optimal climate policy mix? Defining the concept of optimality, including political and legal framework conditions. CECILIA2050 WP1 Deliverable 1.1*. Berlin. Available at: www.cecilia2050.eu.

Görlach, B., Mehling, M. and Roberts, E. (2015) *Designing Institutions, Structures and Mechanisms to Facilitate the Linking of Emissions Trading Schemes*. Berlin.

Goulder, L. *et al.* (2017) ‘China’s national carbon dioxide emission trading system: An introduction’, *Economics of Energy & Environmental Policy*, 6(2).

Goulder, L. H. and Parry, I. W. H. (2008) *Instrument Choice in Environmental Policy*. Available at: www.rff.org.

Goulder, L. H. and Pizer, W. A. (2006) *The Economics of Climate Change*. Available at: www.rff.org.

Goulder, L. H. and Schein, A. (2013) *Carbon taxes vs. Cap and Trade: a critical review*. 19338. Cambridge.

Green, J. F., Sterner, T. and Wagner, G. (2014) 'A balance of bottom-up and top-down in linking climate policies', *Nature Climate Change*.

Grosjean, G., Acworth, W. and Flachsland, C. (2014) 'Envecon 2014 After Monetary Policy , Climate Policy : Is Delegation the Key to EU ETS reform ?', *Climate Policy*, 14(February 2015), pp. 1–16.

Grottera, C. (2013) *Impactos de políticas de redução de emissões de gases do efeito estufa sobre a desigualdade de renda no Brasil*. Universidade Federal do Rio de Janeiro.

Grüll, G. and Taschini, L. (2011) 'Cap-and-trade properties under different hybrid scheme designs', *Journal of Environmental Economics and Management*.

Gunningham, N. and Sinclair, D. (1998) 'Designing smart regulation', *Designing Environmental Policy*, p. 19 p.

Gurgel, A. C. (2012) *Modelo de Equilíbrio Geral*.

Gurgel, A. C. and Paltsev, S. (2014) 'Costs of reducing GHG emissions in Brazil', *Climate Policy*.

GVces (2013) *Elementos e para a Construção de um Sistema de Comércio de Emissões - Volume III*.

GVces (2015) *Implementação do Plano Indústria de baixo carbono: propostas de fomento para a eficiência energética na indústria*. São Paulo.

Haites, E. (2014) *Lessons Learned from Linking Emissions Trading Systems: General Principles and Applications*. Washingto, DC.

Hamdi-Cherif, M., Guivarch, C. and Quirion, P. (2011) 'Sectoral targets for developing countries: Combining "common but differentiated responsibilities" with "meaningful participation"', *Climate Policy*.

Hardin, G. (1968) 'The Tragedy of the Commons Author (s): Garrett Hardin Published by : American Association for the Advancement of Science Stable URL : http://www.jstor.org/stable/1724745', *Advancement Of Science*, 162(3859), pp. 1243–1248.

Harrington, W., Morgenstern, R. D. and Sterner, T. (2004) *Choosing Environmental Policy Comparing Instruments and Outcomes In the United States and Europe*.

Harris, J. M., Roach, B. and Codur, A.-M. (2017) *The Economics of Global Climate Change A GDAE Teaching Module on Social and Environmental Issues in Economics*. Available at: http://ase.tufts.edu/gdae.

Harrod, R. P. and Martin, D. L. (2014) 'The Science of Climate Change'.

Haug, C., Frerk, M. and Santikarn, M. (2015) *Towards a Global Price on Carbon: Pathways for Linking Carbon Pricing Instruments*. Berlin.

Hawkins, S. and Jegou, I. (2014) 'Linking Emissions Trading Schemes Considerations and Recommendations for a Joint EU-Korean Carbon Market', *ICTSD Global Platform on Climate Change, Trade and Sustainable Energy*, (3).

Heindl, P. and Voigt, S. (2012) *Employment effects of regional climate policy: The case of renewable energy promotion by feed-in tariffs*, *ZEW Discussion Papers*.

Heitzig, J. (2013) *Bottom-Up Strategic Linking of Carbon Markets: Which Climate Coalitions Would Farsighted Players Form?*

Helm, C. (2003) 'International emissions trading with endogenous allowance choices', *Journal of Public Economics*, 87(12), pp. 2737–2747.

Hepburn, C. *et al.* (2006) 'Auctioning of EU ETS Phase II Allowances: how and why?', *Climate Policy*, pp. 137–160.

Hoffman, M. J. (2009) 'A Question of Balance: Weighing the Options of Global Warming Policies - By William Nordhaus', *Governance*.

Holtsmark, K. and Midttømme, K. (2016) *The dynamics of linking permit markets*. Available at: http://uk.reuters.com/assets/print?aid=UKL3N0R107420140831.

Hotelling, H. (1931) 'The Economics of Exhaustible Resources Author', *Journal of Political Economy*, 39(2), pp. 137–175.

Hübler, M. (2012) 'Carbon tariffs on Chinese exports: Emissions reduction, threat, or farce?', *Energy Policy*.

Hübler, M., Voigt, S. and Löschel, A. (2014) 'Designing an emissions trading scheme for China-An up-to-date climate policy assessment', *Energy Policy*.

ICAP (2018) *Emissions Trading Worldwide: Status Report 2018*. Berlin.

IEA (2005) *Act Locally, Trade Globally: Emissions trading for climate policy*. OECD.

IEA (2012) *World Energy Outlook. International Energy Agency*. Paris, France. Available at: http://www.worldenergyoutlook.org/publications/weo-2012/.

IETA (2016) *A vision for the Market Provisions of the Paris Agreement*.

IPCC (2001) *Climate Change 2001: Mitigation: Contribution of Working Group III to the Third Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge (UK) and New York (USA).

IPCC (2014) *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Geneva, Switzerland.

Jacoby, H. D. and Ellerman, A. D. (2002) *The Safety Valve and Climate Policy*. Cambridge (USA).

Jaffe, A. B., Newell, R. G. and Stavins, R. N. (2003) 'Technological Change and the Environment', in Mäler, K. G. and Vincent, J. (eds) *Handbook of Environmental Economics*. 1st edn. Amsterdam and Boston: Elsevier Science B.V, pp. 461–516.

Jaffe, A. B. and Stavins, R. N. (1995) 'Dynamic incentives of environmental regulations: The effects of alternative policy instruments on technology diffusion', *Journal of Environmental Economics and Management*, 29(3), pp. S43–S63.

Jaffe, J., Ranson, M. and Stavins, R. N. (2009) 'Linking Tradable Permit Systems: A Key Element of Emerging International Climate Policy Architecture, 36 Ecology', *Ecology Law Quarterly*, 36(4).

Jaffe, J. and Stavins, R. N. (2007) *Linking Tradable Permit Systems for Greenhouse Gas Emissions: Opportunities, Implications, and Challenges*.

Jakob, M. *et al.* (2012) 'Time to act now? Assessing the costs of delaying climate measures and benefits of early action', *Climatic Change*, 114(1), pp. 79–99.

Jegou, I. and Rubini, L. (2011) *The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations ICTSD Global Platform on Climate Change, Trade and Sustainable Energy Transition to a Low Carbon Future Series ICTSD Programme on Competitiveness and Sustainable Devel*. Available at: www.ictsd.org.

Jorgenson, D. W. *et al.* (2008) *White Paper-Pew Center on Global Climate Change The Economic Costs of a Market-based Climate Policy With contributions from Prepared for the Pew Center on Global Climate Change*.

Jotzo, F. and Betz, R. (2009) 'Australia's emissions trading scheme: Opportunities and obstacles for linking', *Climate Policy*.

Kachi, A. *et al.* (2015) *Linking Emissions Trading Systems: A Summary of Current Research*.

Kallis, G., Gómez-Baggethun, E. and Zografos, C. (2013) 'To value or not to value? That is not the question', *Ecological Economics*, (94), pp. 97–105.

Keohane, N. O., Revesz, R. L. and Stavins, R. N. (1998) 'The choice of regulatory instruments in environmental policy', *Harvard Environmental Law Review*, pp. 313–367.

Klepper, G. and Petterson, S. (2006) 'Emissions Trading, CDM, JI, and More: The Climate Strategy of the EU', *The Energy Journal*, 27(2), pp. 1–26.

Kober, T. *et al.* (2014) 'Macroeconomic impacts of climate change mitigation in Latin America: A cross-model comparison', *Energy Economics*.

Kollenberg, S. and Taschini, L. (2016) *Emissions trading systems with cap adjustments*. Available at: http://www.lse.ac.uk/grantham.

Laing, T. *et al.* (2013) *Assessing the effectiveness of the EU Emissions Trading System*. Available at: http://www.lse.ac.uk/grantham.

Lanzi, E. *et al.* (2013) 'Addressing Competitiveness and Carbon Leakage

Impacts Arising from Multiple Carbon Markets: a modelling assessment’.

Lazarowics, M. (2009) *Global Carbon Trading A framework for reducing emissions*. UK. Available at: www.tsoshop.co.uk.

Legro, S. *et al.* (1999) *Climate Change Policy and Programs in Russia: An Institutional Assessmen*. Washingto, DC.

Lima, C. Z. de (2017) *Impacts of low carbon agriculture in Brazil: a CGE application*. Universidade de Viçosa, MG.

Lohmann, L. *et al.* (2006) ‘Carbon trading: a critical conversation on climate change, privatization and power.’, *Development Dialogue*, 1(48), pp. 1–359.

Lohmann, L. (2009) ‘Toward a different debate in environmental accounting: The cases of carbon and cost-benefit’, *Accounting, Organizations and Society*.

Lucena, A. F. P. *et al.* (2014) ‘Climate policy scenarios in Brazil: A multi-model comparison for energy’, *Energy Economics*.

Luderer, G. *et al.* (2012) ‘On the regional distribution of mitigation costs in a global cap-and-trade regime’, *Climatic Change*, 1114(1), pp. 59–78.

MAF (2011) *A guide to forestry in the Emissions Trading Scheme Ministry of Agriculture and Forestry A guide to*. Wellington, New Zealand. Available at: www.maf.govt.nz.

Magalhães, A. S. (2013) *Economia de baixo carbono no Brasil: alternativas de políticas e custos de redução de emissões de gases de efeito estufa*. Universidade Federal de Minas Gerais.

Manzoni, M. (2013) *Elementos para a construção de um sistema de comércio de emissões (volume III)*.

Marschinski, R., Flachsland, C. and Jakob, M. (2012) ‘Sectoral linking of carbon markets: A trade-theory analysis’, *Resource and Energy Economics*.

Marshall, A. (1890) *Principles of Economics*. London: Macmillan.

Massetti, E. and Tavoni, M. (2012) ‘A developing Asia emission trading scheme (Asia ETS)’, *Energy Economics*.

Mathiesen, L. (1985) ‘Computation of economic equilibrium by a sequence of linear complementarity problems’, *Mathematical Programming Study*, 23, pp. 144–162.

McKinsey & Company (2009) *Pathways to a Low Carbon Economy*.

MCTI (2013) *Plano Setorial de Mitigação à Mudança do Clima para Consolidação de uma Economia de Baixa Emissão de Carbono na Indústria de Transformação*. Brasília. Available at: http://www.mma.gov.br/clima/.

MCTI (2014) *Estimativas Anuais de Emissões de Gases de Efeito Estufa no Brasil*. 2nd edn.

MCTI (2018) *Portal SIRENE*. Available at: http://sirene.mcti.gov.br (Accessed: 14 June 2018).

Mehling, M. (2007) ‘Bridging the Transatlantic Divide: Legal Aspects of a Link

Between Regional Carbon Markets in Europe and the United States.', *Sustainable Development Law and Policy*, 7(2), p. 46.

Mehling, M. *et al.* (2011) *Prospects for a Transatlantic Carbon Market: What Next after the US Midterm Elections?* Available at: www.climatestrategies.org.

Mehling, M. (2016) 'Legal Frameworks for Linking National Emissions Trading Systems', in Carlarne, C., Gray, K., and Tarasofsky, R. (eds) *The Oxford Handbook of International Climate Change Law*. Oxford: Oxford University Press, p. 261-288.

Mehling, M. A., Metcalf, G. E. and Stavins, R. N. (2017) *Linking Heterogeneous Climate Policies (Consistent with the Paris Agreement)*, *Harvard Project on Climate Agreements*.

Mehling, M. and Görlach, B. (2016) *Multilateral Linking of Emissions Trading Systems*.

Mehling, M. and Haites, E. (2009) 'Mechanisms for linking emissions trading schemes', *Climate Policy*.

Merkel, A. and François, H. (2015) *Petersberg Dialogue Call for Climate Action - Joint Statement from Angela Merkel and François Hollande*. Available at: https://www.diplomatie.gouv.fr/en/french-foreign-policy/climate/events/article/petersberg-dialogue-call-for (Accessed: 30 June 2018).

Metcalf, G. E. (2009) 'Market-based Policy Options to Control U.S. Greenhouse Gas Emissions', *Journal of Economic Perspectives*, 23(2), pp. 5–27.

Metcalf, G. E. and Weisbach, D. (2012) 'Linking policies when tastes differ: Global climate policy in a heterogeneous world', *Review of Environmental Economics and Policy*.

Millar, R. *et al.* (2017) 'Emission budgets and pathways consistent with limiting warming to 1.5 °C', *Nature Geoscience*, 10, pp. 741–747.

Moarif, S. and Rastogi, N. P. (2012) *Market-based climate mitigation policies in emerging economies*.

Moore, S. (2013) *If the cap fits: reform of the european climate policy and the EU emissions trading system*. Edited by G. Newey. London: Policy Exchange.

Mountondo, E. (1999) 'The Polluter Pays Principle', *EGM Consult*.

Müller, B. and Müller-Fürstenberger, G. (2003) 'Price-related sensitivities of greenhouse gas intensity targets', *Climate Policy*, 3(Supplement 2), pp. S59–S74.

Munnings, C. *et al.* (2016) 'Assessing the design of three carbon trading pilot programs in China', *Energy Policy*. Elsevier, 96, pp. 688–699.

Narassimhan, E. *et al.* (2017) *Carbon-Pricing-In-Practice-A-Review-of-the-Evidence*. Medford, MA.

Narayanan, B., Hertel, T. and Walmsley, T. (2012) *GTAP 8 Data Base Documentation. Chapter 1: Introduction*.

Nascimento, V. M., Nascimento, M. and Bellen, H. M. Van (2013) ‘Instrumentos de políticas públicas e seus impactos para a sustentabilidade’, *Gestão & Regionalidade*, 29(86), pp. 77–87.

Nepstad, D. *et al.* (2014) ‘Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains’, *Science*, 344(6), pp. 1118–1123.

Newell, P. and Paterson, M. (2013) *The Politics of the Carbon Economy*.

Newell, R. G., Pizer, W. A. and Raimi, D. (2013) *Carbon Markets: Past, Present, and Future*. Available at: www.rff.org.

Newell, R. G. and Stavins, R. N. (2003) ‘Cost Heterogeneity and the Potential Savings from Market-Based Policies’, *Journal of Regulatory Economics*.

Nicolletti, M. and Lefévre, G. B. (2016) ‘Precificação de carbono no Brasil: perspectivas e aprendizados a partir de uma simulação de mercado cap-and-trade’, *FGV EAESP - GVces*, pp. 145–169. Available at: http://bibliotecadigital.fgv.br/dspace/bitstream/handle/10438/18725/GVces_Nicolletti%3B Lefevre.pdf?sequence=1&isAllowed=y.

Nordhaus, W. (2015) ‘Climate clubs: Overcoming free-riding in international climate policy’, *American Economic Review*.

Nordhaus, W. D. and Boyer, J. (1999) ‘Requiem for Kyoto: An Economic Analysis of the Kyoto Protocol’, *The Energy Journal*. The Energy Journal, (The Costs of the Kyoto Protocol: A Multi-Model Evaluation), pp. 93–130.

Oates, W. and Baumol, W. (1988) *The Theory of Environmental Policy*. 2nd edn. Cambridge University Press.

OCDE (1989) *Economic instruments for environmental protection*. Paris.

Octaviano, C., Paltsev, S. and Gurgel, A. C. (2014) ‘Climate change policy in Brazil and Mexico: Results from the MIT EPPA model’, *Energy Economics*.

Ojha, V. (2011) ‘Carbon emissions reduction strategies and poverty alleviation in India’, *Environment and Development Economics*, 14, pp. 323–348.

Page, E. A. (2011) ‘Cashing in on climate change: Political theory and global emissions trading’, *Critical Review of International Social and Political Philosophy*.

Paltsev, S. *et al.* (2005) ‘The MIT, Emissions Prediction and Policy Analysis (EPPA) model version 4’, *MIT joint program on the science and policy of global change*, Report No(125), pp. 1–73.

Paltsev, S. *et al.* (2012) ‘The role of Chinage in mitigating climate change’, *Energy Economics*, 34, pp. S444–S450.

Pearce, D. W. and Turner, R. K. (1990) *Economics of Natural Resources and the Environment*. Baltimore: John Hopkins University Press.

Peñasco, C. and Río, P. Del (2015) *Dialogue on a RES policy framework for 2030 About the project Who we are?* Available at: www.towards2030.eu.

Perman, R. *et al.* (2011) *Natural Resource and Environmental Economics*. Third. Edinburgh Gate, Harlow, Essex: Pearson Education Limited. Available at: www.booksites.net.

Petsonk, A., Keohane, N. O. and Samans, R. (2015) *Creating a Club of Carbon Markets: Implications of the Trade System*. Geneva. Available at: www.e15initiative.org/.

Philibert, C. and Reinaud, J. (2004) *Emissions Trading : Taking Stock and Looking Forward*. Paris, France.

Pigou, A. C. (1920) *The Economics of Welfare*. London: Macmillan.

Pizer, W. A. (1999) *Choosing price or quantity controls for greenhouse gases*, *Climate Issues Brief*. Washingto, DC.

Pizer, W. A. (2002) 'Combining price and quantity controls to mitigate global climate change', *Journal of Public Economics*.

Pizer, W. A. and Yates, A. J. (2015) 'Terminating links between emission trading programs', *Journal of Environmental Economics and Management*.

PMR and ICAP (2016) *Emissions Trading in Practice: a Handbook on Design and Implementation*. Washington, DC.

Poloncarz, K. (2017) 'For richer, for poorer: the California, Ontario, and Québec carbon market', *Section of Environment, Energy, and Resources of the American Bar Association*, 49(5).

Posner, E. A. and Weisbach, D. (2010) *Climate Change Justice*. Princeton: Princeton University Press.

Prag, A., Hood, C. and Barata, P. (2013) *Made to Measure: Options for Emissions Accounting under the UNFCCC*. Available at: www.oecd.org/env/cc/ccxg.htmwww.iea.org.

Purdon, M., Houle, D. and Lachapelle, E. (2014) *The Political Economy of California and Québec's Cap-and-Trade Systems*. Otawa. Available at: www.sustainableprosperity.ca.

Qi, T. *et al.* (2013) *The Energy and Economic Impacts of Expanding International Emissions Trading TSINGHUA-MIT The Energy and Economic Impacts of Expanding International Emissions Trading*.

Qi, T. *et al.* (2015) 'An analysis of China's climate policy using the China-in-Global Energy Model', *Economic Modelling*, 52, pp. 650–660.

Quemin, S. and De Perthuis, C. (2017) 'Transitional Restricted Linkage between Emissions Trading Schemes'. Available at: www.faere.fr.

Ranson, M. and Stavins, R. (2012) 'Post-Durban Climate Policy Architecture Based on Linkage of Cap-and-Trade Systems', *The Chicago Journal of International Law*, 13(2), pp. 403–438.

Ranson, M. and Stavins, R. N. (2013) *Linkage of Greenhouse Gas Emissions Trading Systems Learning from Experience Linkage of Greenhouse Gas Emissions Trading Systems: Learning from Experience Robert N. Stavins*

Harvard Project on Climate Agreements. Washingto, DC.

Rathmann, R. (2012) *Impactos da adoção de metas de redução de emissão gases de efeito estufa sobre a competitividade de setores industriais energointensivos do Brasil*. Universidade Federal do Rio de Janeiro.

Rausch, S., Metcalf, G. E. and Reilly, J. M. (2011) *Distributional impacts of carbon pricing: a general equilibirum approach with micro-date for households*. 17087. Cambridge.

Reilly, J. M. *et al.* (2007) 'Assessment of U.S. Cap-and-Trade Proposals'.

Revelle, E. (2009) 'Cap-and-trade versus carbon tax two approaches to curbing Greenhouse gas emissions', *LWVUS Climate Change Task Force*.

Rezek, J. and Blair, B. . (2005) 'Abatement Cost Heterogeneity in Phase I Electric Utilities', *Contemporary Economic Policy,* 23(3), pp. 324–340.

Riahi, K., Grübler, A. and Nakicenovic, N. (2007) 'Scenarios of long-term socio-economic and environmental development under climate stabilization', *Technological Forecasting and Social Change*, 74(7), pp. 887–935. Available at: https://ac.els-cdn.com/S0040162506001387/1-s2.0-S0040162506001387-main.pdf?_tid=9e8c861c-94af-45fe-84b6-11c18b5ba53e&acdnat=1535720488_ac109bf0a50e17648d06fb4e25e7d093.

Romeiro, A. (2012) 'Sustainable development: an ecological economics perspective', *Estudos Avançados*, 26(74).

La Rovere, E. L. (2016) 'O Brasil e a COP-21', in *Mudanças climáticas: o desafio do século*. Rio de Janeiro: Fundação Konrad Adenauer, pp. 11–24.

Rusche, T. M. (2010) 'The European climate change program: An evaluation of stakeholder involvement and policy achievements', *Energy Policy*. Elsevier, 38(10), pp. 6349–6359.

Russell., C. and Powell, P. (1996) *Choosing Environmental Policy Tools Theoretical and Practical Considerations.* Washingto, DC.

Rutherford, T. F. (1995) 'Extension of GAMS for complementarity problems arising in applied economic analysis', *Journal of Economic Dynamics and Control*, 19(8), pp. 1299–1324.

Rutherford, T. F. (1999) 'Applied general equilibrium modeling with MPSGE as a GAMS subsystem: an overview of the modeling framework and syntax', *Computational economics Kluwer academic publishers*, 14, pp. 1–46.

Rydge, J. (2015) *Implementing Effective Carbon Pricing. Contributing paper for Seizing the Global Opportunity: Partnerships for Better Growth and a Better Climate*. London and Washington, DC. Available at: http://newclimateeconomy.report/misc/working-papers/.

Rydge, J. (no date) *Implementing Effective Carbon Pricing*. Available at: www.newclimateeconomy.report.

Sartor, O. *et al.* (2015) *What does the European power sector need to decarbonise? The role of the EU ETS & complementary policies post-2020*.

Sawyer, D., Peters, J. and Stiebert, S. (2016) ‘Overview of Macroeconomic and Household Impacts of Ontario’s Cap and Trade Program’, 2016. Available at: http://www.enviroeconomics.org/#!Impact-Modelling-and-Analysis-of-Ontario’s-Proposed-Cap-and-Trade-Program/c1uze/573a64620cf23f57cc66dd05.

Schmalensee, R. and Stavins, R. N. (2015) ‘Lessons learned from three decades of experience with Cap-and-Trade’, *Review of Environmental Economics and Policy*.

Schmalensee, R. and Stavins, R. N. (2017a) ‘Lessons learned from three decades of experience with cap and trade’, *Review of Environmental Economics and Policy*.

Schmalensee, R. and Stavins, R. N. (2017b) ‘The design of environmental markets: What have we learned from experience with cap and trade?’, *Oxford Review of Economic Policy*.

Schneider, L. *et al.* (2017) ‘Restricted Linking of Emissions Trading Systems: Options, Benefits, and Challenges’, *International Environmental Agreements*, 17, pp. 1–16.

SEEG (2016) *Análise das emissões de GEE Brasil (1970-2014) e suas implicações para políticas públicas e contribuição brasileira para o Acordo de Paris*. Available at: http://seeg.eco.br/wp-content/uploads/2016/09/WIP-16-09-02-RelatoriosSEEG-Sintese.pdf.

Seroa da Motta, R. (1996) *Indicadores Ambientais no Brasil: Aspectos Ecológicos, de Eficiência e Distributivos*. Rio de Janeiro.

Seroa da Motta, R. (2004) *Analyzing the environmental performance of the Brazilian industrial sector*. 1053. Rio de Janeiro.

Seroa da Motta, R. (2006) *Economia ambiental*. Rio de Janeiro: FGV.

Seroa da Motta, R. (2011) ‘A Política nacional sobre mudança do clima: aspectos regulatórios e de governança’, in Seroa da Motta, R. et al. (eds) *Mudança do Clima no Brasil: aspectos econômicos, sociais e regulatórios*. Brasília: Instituto de Pesquisa Econômica Aplicada (IPEA), pp. 31–42.

Seroa da Motta, R., Couto, L. C. and Castro, L. (2012) *Curvas de Custos Marginais de Abatimento de Gases de Efeito Estufa no Brasil: Resenha e Oportunidades de Mitigação*. Brasília.

Seroa da Motta, R., Ruitenbeek, J. and Huber, R. (1996) ‘Uso de instrumentos econômicos na gestão ambiental da América Latina e Caribe: lições e recomendações’, *IPEA - Texto para discussão*, nº 440, p. 66.

Sirkis, A. (2016) ‘Avaliando a COP-21: quão cheio ficou o copo?’, *Revista ECO21*, 230. Available at: http://www.eco21.com.br/textos/textos.asp?ID=3774.

Somanathan, E. (2008) *What Do We Expect from an International Climate Agreement? A Perspective from a Low-income Country The Harvard Project on International Climate Agreements*. Available at: www.belfercenter.org/climate.

Soto, J. (2016) ‘A direção está mais do que clara na COP-21’, *Revista ECO21*. Available at: http://www.eco21.com.br/textos/textos.asp?ID=3772.

Stavins, R. *et al.* (2014) ‘International Cooperation: Agreements and Instruments’, in Edenhofer, O. et al. (eds) *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. London (UK) and New York (USA): Cambridge University Press,.

Stavins, R. N. (1997a) *Economic incentives for environmental regulation*.

Stavins, R. N. (1997b) *Policy Instruments for Climate Change: How Can National Governments Address a Global Problem*, *University of Chicago Legal Forum*. Available at: http://heinonline.org/HOL/Page?handle=hein.journals/uchclf1997&id=297&div=&collection=journals%5Cnhttp://www.hks.harvard.edu.proxy.library.vanderbilt.edu/fs/rstavins/Papers/Policy Instruments for Climate Change.pdf.

Stavins, R. N. (2008) ‘Cap-and-Trade or a Carbon Tax? (methods for Controlling Greenhouse Gases)’, *The Environmental Forum*, 25. Available at: https://scholar.harvard.edu/files/stavins/files/column_22.pdf.

Stavins, R. N. (2015) *Linkage of regional, national, and sub-national policies in a future international climate agreement*.

Sterk, W. *et al.* (2006) *Joint Emissions Trading as a Socio-Ecological Transformation Cross-Section Project 4 Ready to Link Up? Implications of Design Differences for Linking Domestic Emissions Trading Schemes*.

Sterk, W., Mehling, M. and Tuerk, A. (2009) ‘Prospects of Linking EU and US Emission Trading Schemes: Comparing the Western Climate Initiative, the Waxman-Markey and the Lieberman-Warner Proposals’, *Climate Strategies*.

Sterk, W. and Mersmann, F. (2011) *Sterk and Mersmann Domestic Emission Trading Systems in Developing Countries-State of Play and Future Prospects Sterk and Mersmann Sterk and Mersmann*.

Stern, N. (2006) ‘The Economics of Climate Change. The Stern Review’, *October*, 30(3), p. 27.

Sterner, T. (2002) *Policy Instruments for Environmental and Natural Resource Management*. Washington, DC: RFF Press (Resources for the Future) and World Bank.

Sterner, T. (2003) *Policy Instruments for Environmental and Natural Resource Management*. Edited by Resources for the Future.

Studart, R. and Gallagher, K. (2016) ‘Guaranteeing Finance for Sustainable Infrastructure: a Proposal’, in Sirkis, A. et al. (eds) *Moving the trillions - a debate on positive pricing of mitigation actions*. Rio de Janeiro, pp. 91–112.

Taschini, L. (2009) *Environmental economics and modeling marketable permits*. 34 and 25. UK.

Taschini, L. (2011) ‘Pollution permits , strategic trading and dynamic technology adoption Santiago Moreno-Bromberg and Luca Taschini May 2011 Centre for Climate Change Economics and Policy Working Paper No . 57 Grantham Research Institute on Climate Change and the Environmen’, (57).

Taschini, L., Fankhauser, S. and Hepburn, C. (2011) *The role of auctions in promoting GHG abatement in emerging markets*.

Tavoni, M. and Socolow, R. (2013) 'Modeling meets science and technology: An introduction to a special issue on negative emissions', *Climatic Change*, 118(1), pp. 1–14.

The Royal Society (2002) *Economic instruments for the reduction of carbon dioxide emissions*. 26/02. Available at: www.royalsoc.ac.uk.

Tiche, F. G., Weishaar, S. E. and Couwenberg, O. (2016) *Carbon Market Stabilisation Measures: Implications for Linking*.

Tietenberg, T. (1985) *Emissions trading : an exercise in reforming pollution policy*. second. Washington, DC: RFF Press (Resources for the Future).

Tietenberg, T. (2003) 'The Tradable-Permits Approach to Protecting the Commons: Lessons for Climate Change', *Oxford Review of Economic Policy*, 19(3), pp. 400–419.

Tol, R. *et al.* (2004) 'Distributional aspects of climate change impacts', *Global Environmental Change*, 14, pp. 259–272.

Tol, R. S. J. (2009) 'The Economic Effects of Climate Change', *Journal of Economic Perspectives*, 23(2), pp. 29–51.

Tol, R. S. J. (2015) *Economic impacts of climate change*. Sussex.

Tol, R. S. J. (2018) 'The Economic Impacts of Climate Change', *Review of Environmental Economics and Policy*, 12(1), pp. 2–25.

Tomas, M. J. and Callan, S. J. (2010) *Economia ambiental: aplicações, políticas e teorias*. São Paulo: Lenguage Learning.

Tuerk, A. *et al.* (2009) *Linking Emissions Trading Schemes: Synthesis Report*.

Tvinnereim, E. and Mehling, M. (2018) 'Carbon pricing and deep decarbonisation', *Energy Policy*, 121, pp. 185–189.

UNCTAD (2016) *Virtual institute teaching material on trade, the environment and sustainable development: transition to a low-carbon economy*. New York and Geneva.

UNEP (2004) *The Use of Economic Instruments for Environmental Policy: Opportunities and Challenges*. UNEP.

UNFCCC (1992) *Text of the Convention*.

UNFCCC (2015a) *Paris Agreement*. Available at: https://unfccc.int/sites/default/files/english_paris_agreement.pdf.

UNFCCC (2015b) *Synthesis report on the aggregate effect of intended nationally determined contributions.* Available at: http://unfccc.int/focus/indc_portal/items/9240.php (Accessed: 31 May 2018).

Verde, S. F. and Tol, R. S. J. (2009) 'The Distributional Impact of a Carbon Tax in', *The Economic and Social Review*, 40(3), pp. 317–338.

Victor, D. G. (2015) *Strengthening the Global Trade System E The Case for*

Climate Clubs. Available at: www.e15initiative.org.

Vieira, L. and Vernet, E. (2015) 'COP-21: Sucesso Diplomático, Fracasso Climático', *Revista ECO21*, 229. Available at: http://www.eco21.com.br/textos/textos.asp?ID=3751.

Viola, E. (2004) 'Brazil in the context of global governance politics and climate change, 1989-2003', *Ambiente & sociedade*, 7(1), pp. 27–46.

Wagner, G. (2014) *Linking sound economics with global politics*. Available at: http://gwagner.com/wp-content/uploads/Wagner-Upton-Linkage-141231-gw.pdf (Accessed: 8 June 2018).

Walras, L. (1969) *Elements of pure economics; or, The theory of social wealth*. New York.

Wills, W. and Lefevre, J. (2012) 'The impact of a carbon tax over the Brazilian economy in 2030 - IMACLIM: the hybrid CGE model approach', in *Ecoeco 2012*. Available at: http://www.isecoeco.org/conferences/isee2012-versao3/pdf/p157.pdf.

Wing, I. S. *et al.* (2006) *Absolute vs. Intensity Limits for CO2 Emission Control: Performance Under Uncertainty*.

Wing, I. S. (2011) 'Computable General Equilibrium Models for the Analysis of Economy-Environment Interactions', in Batabyal, A. and Nijkamp, P. (eds) *Research Tools in Natural Resource and Environmental Economics*.

Wood, P. J. and Jotzo, F. (2011) 'Price floors for emissions trading', *Energy Policy*.

World Bank (2014) *State and Trends of Carbon Pricing 2014*. Washingto, DC.

World Bank (2018) *World Bank Open Data*. Available at: https://data.worldbank.org/ (Accessed: 17 May 2018).

World Bank, Ecofys and Vivid Economics (2017) *States and Trends of Carbon Pricing 2017*. Washingto, DC.

WTO-UNEP (2009) *Trade and Climate Change*. Switzerland.

Xu, Z. *et al.* (2017) 'The role of multi-region integrated emissions trading scheme: A computable general equilibrium analysis q'.

Yu, V. P. B. (2009) *Developing Country Perspectives on Carbon-Based Competitiveness, Trade and Climate Change Linkages*, *Energy, Environment and Development Programme Paper*. London.

Zeng, Y., Weishaar, S. E. and Couwenberg, O. (2016) 'Absolute vs. Intensity-based Caps for Carbon Emissions Target Setting: A Risk Linking the EU ETS to the Chinese National ETS?', *European Journal of Risk Regulation*, 7(4), pp. 764–781.

Zetterberg, L. (2012) *Linking the Emissions Trading Systems in EU and California*. Stockholm.

Zetterberg, L. *et al.* (2012) 'Short-run allocation of emissions allowances and long-term goals for climate policy', *Ambio*.

ZEW (2016) *Annual Report 2016: the year in retrospect*.

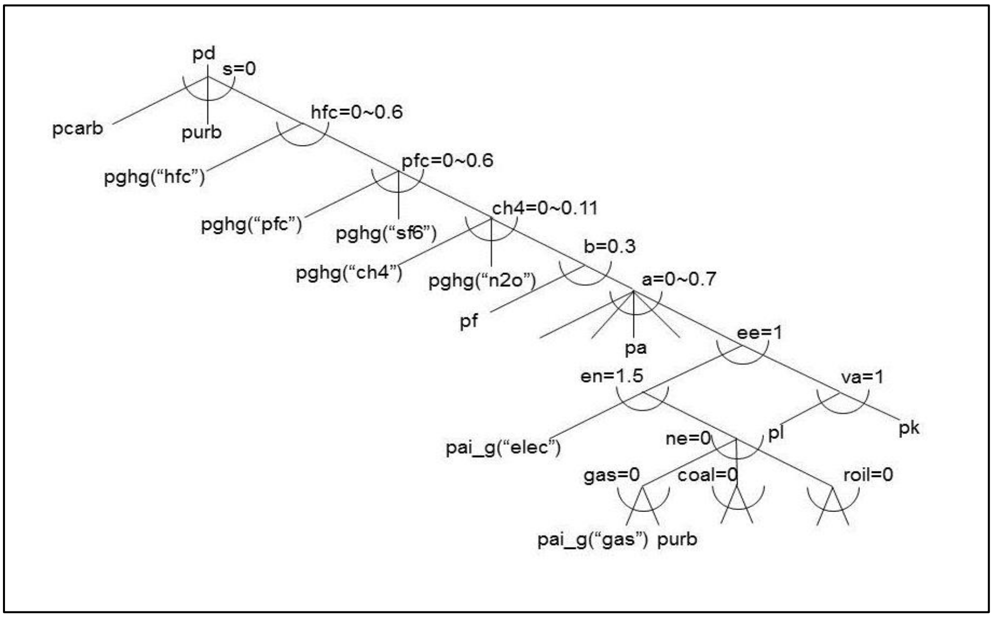
Zhang, D. *et al.* (2014) 'Emissions trading in China: Progress and prospects', *Energy Policy*.

Zhang, Z. X. and Folmer, H. (1998) 'Economic Modelling Approaches to Cost Estimates for the Control of Carbon Dioxide Emissions', *Energy Economics*, 20, pp. 101–120.

van der Zwaan, B., Calvin, K. V. and Clarke, L. E. (2016) 'Climate Mitigation in Latin America: Implications for Energy and Land Use: Preface to the Special Section on the findings of the CLIMACAP-LAMP project', *Energy Economics*.

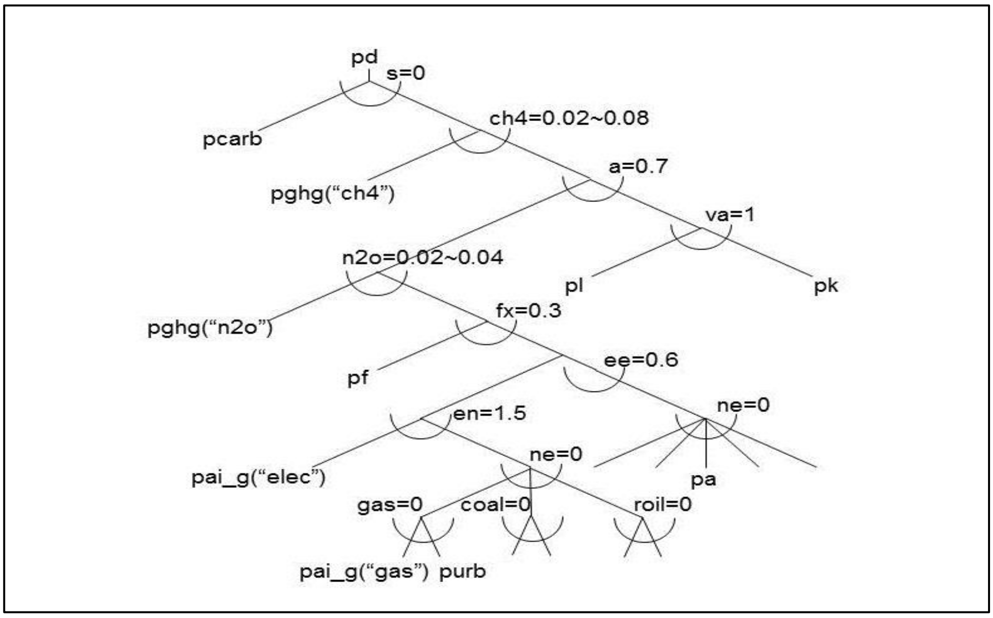
APPENDIX A - OTHER PRODUCTION STRUCTURES IN EPPA6

Figure A1 – Technological structure for FOOD, OTHR, SERV, TRAN, and DWE



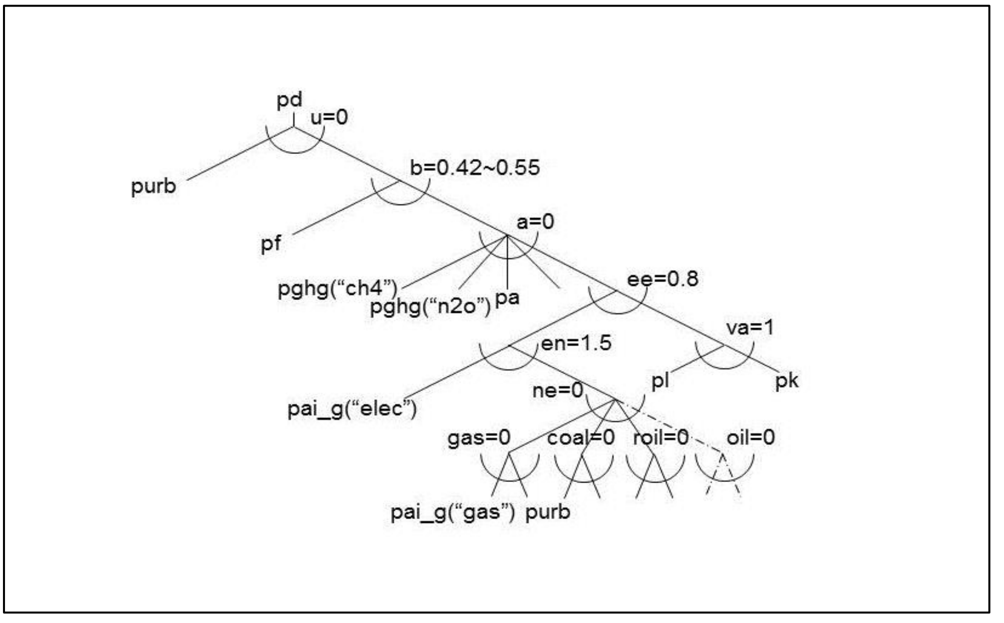
Source: Chen *et al.* (2015).

Figure A2 – Technological structure for CROP, LIVE and FORS



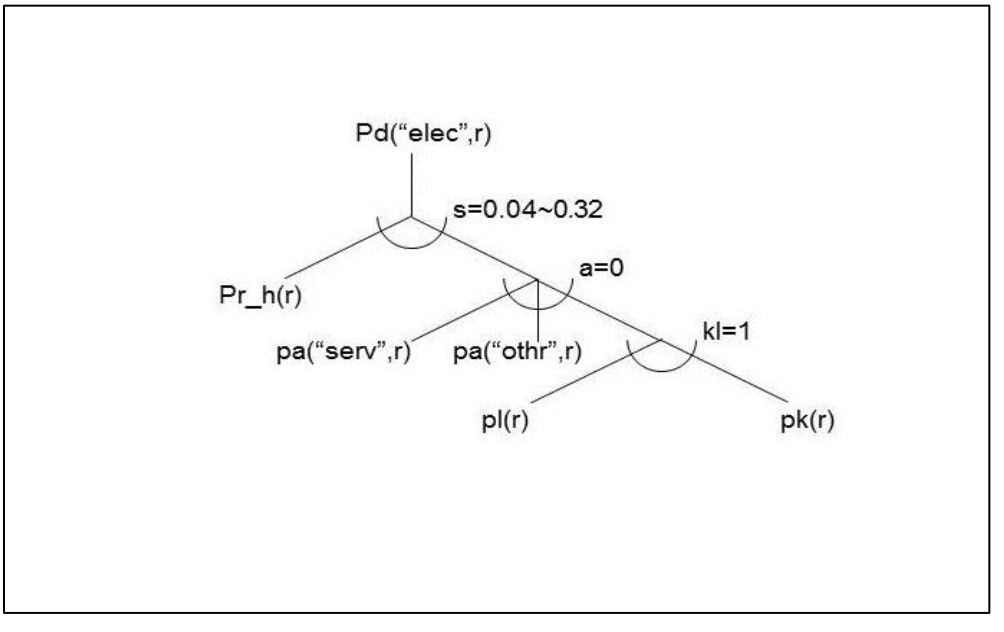
Source: Chen *et al.* (2015).

Figure A3 – Technological structure for COAL, OIL, ROIL and GAS



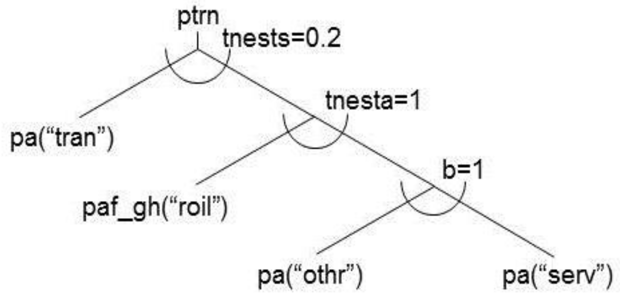
Source: Chen *et al.* (2015).

Figure A4 – Technological structure for HYDRO and NUCLEAR generation



Source: Chen *et al.* (2015).

Figure A5 – Technological structure for household transportation



Source: Chen *et al.* (2015).

Table A1 – Sets in EPPA6

Abbreviation	Description
R	Regions
I, J	Sectors and goods
t	time dimension
NE	non-energy commodities (subset of I)
NEND	non-energy commodities excluding dwe (subset of ne)
dwe	owenership of dwelings (subset of ne)

Table A2 – Productive activities and price index

Abbreviation	Description
w (r)	welfare index
z (r)	consumption index
htrn (r)	household transport
a (g, r)	Armington index
d (g,r)	domestic production index
pu	consumption price index
pw	welfare price index
pt	international transport price
ptrn	household transport price index
pr_h (r)	return to hydro resource factor
pai_c	input price gross of carbon tax - intermediate
paf_c	input price groos of carbon tax - final
pai_g	input price gross of GHG tax - intermediate
paf_g	input price gross of carbon tax - final
paf_gh	input price gross of GHG tax - household transport
pa	Armington price
pd	domestic price
pk	capital price
pf	fixed factor price
pl	wage
pg	government output price
pinv	investment price
pcarb	non-tradable co2 emission permit price
pren	price renewables
pghg	GHG price - national
purb	non-GHG gases

Table A3 – Technological coefficients

Abbreviation	Description
sigg	elasticity of substitution for GHG
sigu	top level transformation elasticity between production and urban gases
enesta	energy input to electricity sector nest a substitution elasticity
tnests	household transport top nest elasticity (between purchased and own-supplied)
tnesta	household transport substitution elasticity between roil and the rest of own-supplied transport
u	Aidads utility level
l_k	labour versus capital
e_kl	labour and capital versus energy bundle
s	specifies elasticity
delas	final demand elasticity between energy and non-energy composites
d_elas	top final demand substitution elasticity
ee	elasticity between energy commodities and added value
va	elasticity between components of added value

APPENDIX B – A DOMESTIC BRAZILIAN ETS UNDER DIFFERENT SCENARIOS

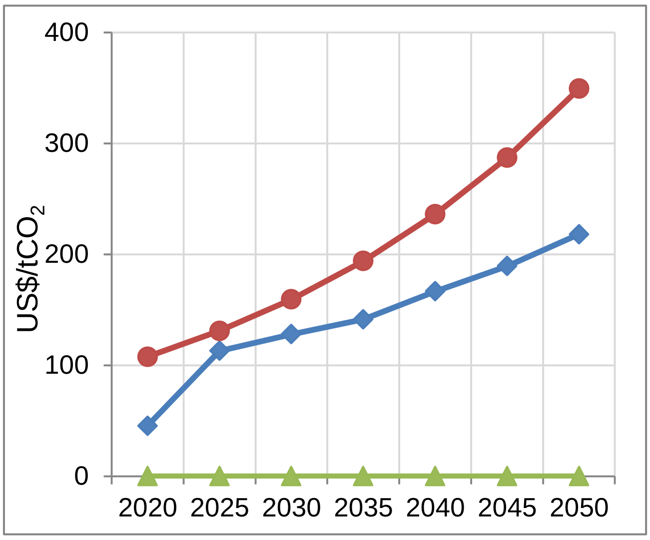
The main objective of this thesis is to analyse environmental and economic implications of linking a proposed sectoral ETS in Brazil to other carbon markets. As such, the ETS setting has been closely aligned to the EU ETS design to potentially facilitate acceptance and negotiations of the bilateral climate agreement. This approach put a greater emphasis on the linkage itself, than on the domestic sectoral ETS. To a certain extent, however, it ends up disregarding domestic abatement opportunities, and the related costs of alternative ETS design options. Of course, in CGE modelling a reduced carbon constraint or sectoral coverage could play an important role in modifying the direction of emissions transfers between sectors and regions, but also the magnitude of GDP and welfare losses.

In this context, we simulated three complementary scenarios for 2020-2050, as summarised in Table B1. Similarly to the link-Ambition scenarios of Chapter 4, in B1 the sectoral ETS is capped with a less stringent mitigation target (which excludes the share of deforestation in total emissions, that is, 27.5%) without flexibility provisions. In the B2 scenario, banking is incorporated in the ETS, which is often used if carbon prices are expected to increase. Finally, scenario B3 implements a Bra-ETS regulating all sectors of the economy, the simplest manner of representing emissions trading in CGE models.

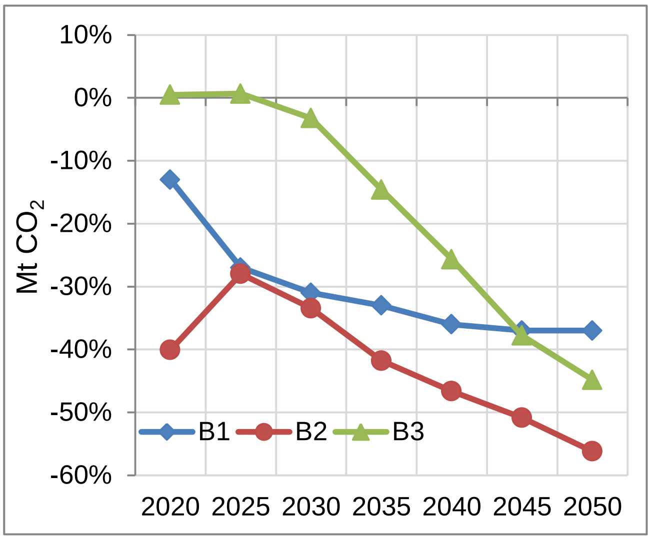
Table B 1 – Summary of alternative scenarios

No.	Scenarios	Description
B1	Bra-ETS-Ambition	a Bra-ETS with reduced mitigation ambition, no banking, no revenue recycling
B2	Bra-ETS-Banking	Bra-ETS that allows only banking
B3	Bra-ETS-All	Bra-ETS for all sectors of the economy

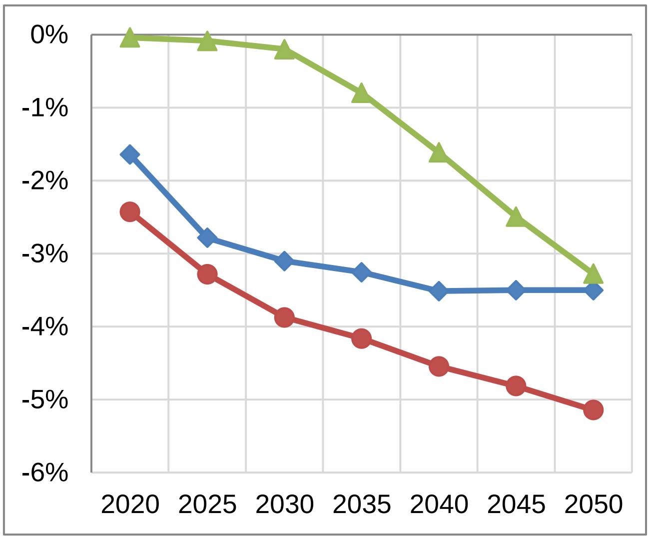
Figure B 1 – Impacts of a Brazilian ETS under alternative scenarios



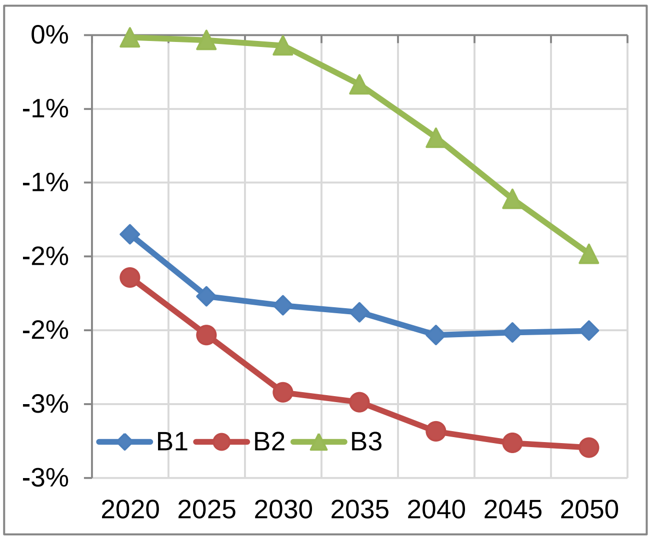
(a) Carbon prices



(b) Sectoral Emissions



(c) GDP



(d) Welfare

Evidences in the Bra-ETS scenarios evaluated in Chapter 4 show that capping emissions of electricity and energy-intensive industries based on the NDC commitment is very costly to Brazil, even though it leads to large emission reductions. Results in Figure B1 indicate that, if a relatively less stringent target is applied, there would be lower GDP and welfare losses over the period of compliance compared to the standard scenario, with and without banking provisions. Alternatively, there are small negative impacts of an ETS designed to cover all sectors of the economy, particularly due to the associated carbon price. Since carbon prices are determined by market forces, results suggest that when all sectors are involved there tends to be an oversupply of permits. In the absence of permit scarcity, the price collapses and incentives to adjust to the target are reduced, which is translated in the number of permits being traded in the carbon market.

In the scenario (Bra-ETS-All), although impacts on welfare and GDP are lower, it restrains energy substitution towards low-carbon technologies. Compared to the other scenarios, the use of coal, gas and oil continue to be widely used compared to other policy scenarios. The sectoral ETS as modelled is more effective in conducting a transition to a low-carbon economy through environmentally friendly technology, notably by increasing the use of renewables (wind and solar). In this sense, the more stringent the target, the greater the substitution effect and the more adverse effects on GDP and welfare.

Since Brazil has been actively participating in international negotiations for addressing climate change, carbon pricing could play a role in helping to decarbonise the economy. In consideration of the Brazilian development level, a sectoral ETS with a less strict cap allows mitigation to take place whilst avoiding further losses in economic activity. Accommodating provisions for banking in this context could assure lower long-term carbon prices to the extent which banked allowances are gradually introduced in the market. In theory, an aggregate ETS serves to internalise the cost of emissions equally across sectors, where those with higher marginal cost abatement has the flexibility to acquire permits to emit in a more liquid market. In practice, however, this proposal is hardly implementable since it is difficult to accurately quantify emissions in most of these sectors. Thus, using the ETS for mitigation on electricity and energy-intensive sectors is more appropriate whereas other strategies are implemented in other sectors.

APPENDIX C – CAPPING THE BRAZILIAN ETS WITH THE INDUSTRIAL PLAN TARGETS: EFFECTS FOR DIFFERENT LINKING SCENARIOS

The results of linking the proposed Brazilian emissions trading strategies to partners of the developed and developing world are presented in Chapter 5. It has been verified that the binding limit on the sectoral ETS in Brazil is set at a relatively high target, which compares to the EU ETS environmental goals. For Brazil, associated costs are substantial without trading and if no additional provisions are included for recycling revenues. In fact, besides reducing the policy costs, this design option would help to increase acceptance of carbon pricing whilst inducing a wider adoption of low-carbon technologies as it makes the final price of this energy artificially less expensive.

In the Paris Agreement, Brazil has explicitly committed to diversify and expand the use of alternative energy. Since market-based instruments have not been specifically mentioned, and assuming that to cap emissions of electricity and energy-intensive industries at the NDC level is ambitious, in this Appendix it is proposed to investigate the emissions trading proposals where the Brazilian ETS is capped with the Industrial Plan target, that is, a 5% reduction per 5-year interval (also equivalent to 35% of emission reductions in 2050). Compared to the mitigation target applied in all simulations in the main part, this is considered to be less strict to regulated sectors. In this case, a higher supply of allowances is expected and hence, lower carbon prices at the domestic level.

As for the linkages, the effects for Brazil depend on the partner's environmental objectives and the existing heterogeneity of costs and abatement opportunities among them. Table C1 summarises the scenarios simulated, which consider the domestic sectoral ETS as well as bilateral linkages and the global linkage. Figure C1 presents economic impacts of sectoral ETS with and without coupling the ETS system with others in terms of carbon price, GDP and welfare changes compared to the No-Policy scenario, as well as the environmental effectiveness of each proposal, that is, the proportion of emission reductions achieved compared to No-Policy scenario.

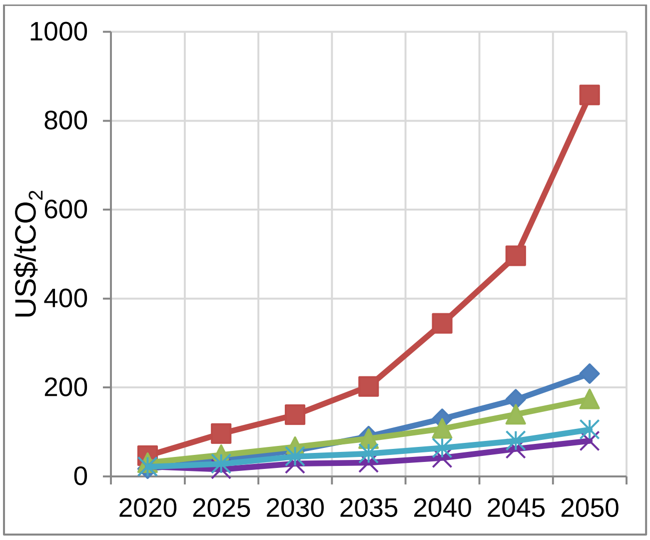
Table C1 – Summary of alternative scenarios

No.	Scenarios
C1	Bra-ETS
C2	Bra-EU-Trade
C3	Bra-LA-Trade
C4	Bra-CHN-Trade
C5	GBL-Trade

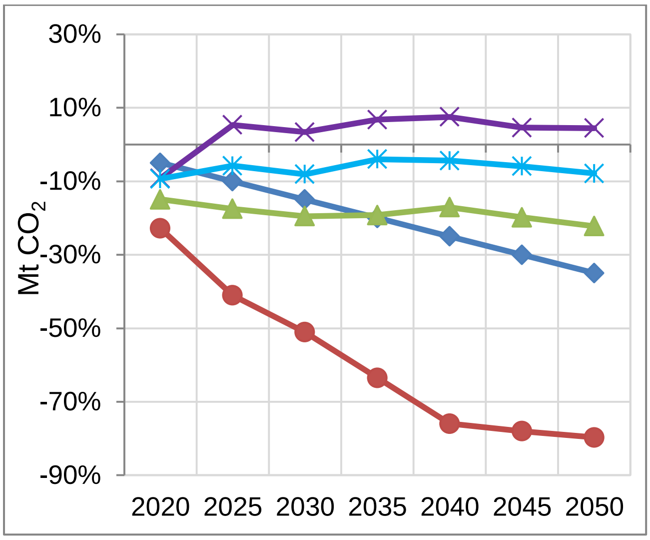
Overall, an upward trend of the carbon price is observed, with a reasonable and stable price rise over time. If linked to the EU ETS system, the carbon cost remains the highest due to the market size and the strict cap which the covered sectors in the EU ETS are subject to. Hence, it is equalised at US$138/t$CO_2$ in 2030 and US$857/t$CO_2$ in 2050. In this simulation, the lower restriction on emissions from electricity and energy intensive sectors in Brazil increases the supply of allowances in the domestic ETS market thereby reducing the carbon price to US$58/t$CO_2$ in 2030 and US$231/t$CO_2$ in 2050, rather than US$202/t$CO_2$ in 2030 and US$305/t$CO_2$ in 2050 in the Bra-ETS scenario. Yet, trading permits with Latin America, China or in the Global ETS enables the Brazilian emitters to adjust to the target more cost-effectively. The Bra-CHN-Trade scenario (C4) has the cheapest allowance cost, followed by the GBL-Trade (C5).

As a result of general equilibrium effects, GDP is negatively impacted the more stringent the partner mitigation level. It is worth noting that there is a drop in GDP in the Bra-EU-Trade compared to the situation where the targets are similarly ambitious, from 3.9% to 2.7% in 2030 and from 5.5% to 3.5% in 2050. In the long-term, linking is GDP improving in all other perspectives, but any effect is verified in terms of welfare. The only exception is the Brazil-Europe linkage, where there are positive gains in welfare involved for Brazil. This occurs because Brazil benefits in linkage by exporting permits to Europe. In parallel, mitigation is the deepest thereby implying that this linkage could be effective in leading to greater abatement without undermining economic development in Brazil. On the other hand, capping those sectors with Industrial Plan targets does not provide a strong signal to curb emissions, particularly if coupled to the Chinese ETS, as it drives emissions up.

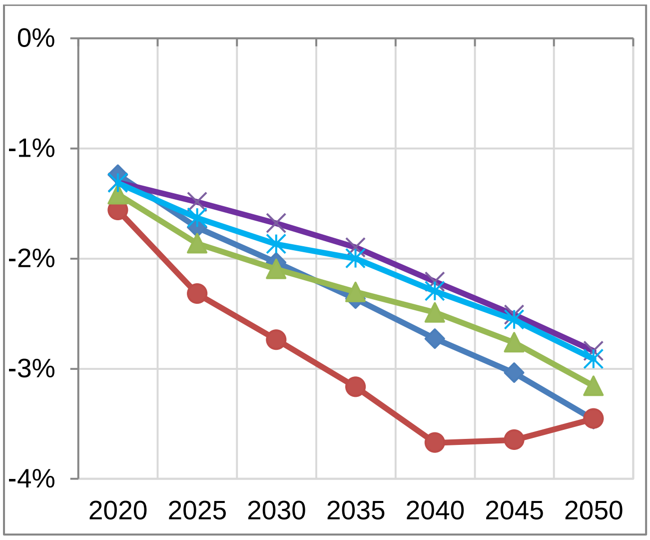
Figure C1 – Impacts of a Brazilian ETS based on the Industrial Plan targets under alternative scenarios



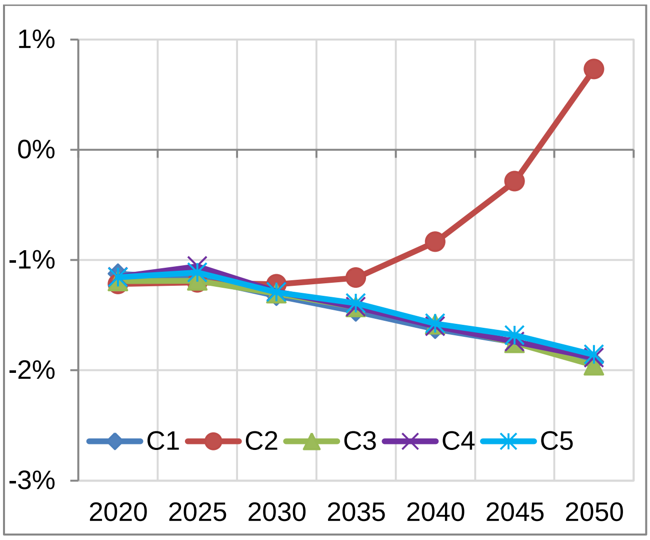
(a) Carbon prices



(b) Sectoral emissions



(c) GDP



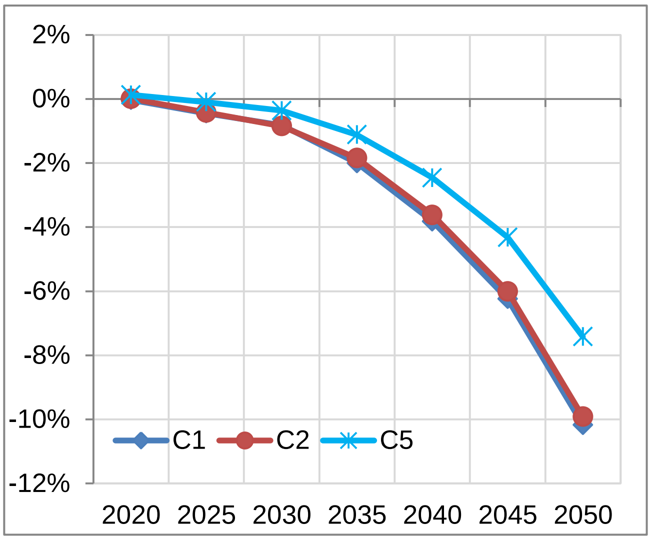
(d) Welfare

On the other hand, there is barely any effect of reducing the Brazilian mitigation target on the GDP of any potential trading partners, as evidenced in Figure C2. Over the period, results highlight that the policy is more costly to Europe and Latin America, with 10% and 6.5% GDP loss in 2050, respectively. Even though there is a downtrend in China’s GDP, the region is barely affected by the carbon policy, whether linked or not, since the target is not effective either to incentivise abatement or to create scarcity of allowances in the market. The Global ETS appears a good strategy to compel China into deeper emission reductions and to trade permits in the market. Europe and Latin America benefit from the global sectoral ETS system by purchasing emissions, as do Brazil.

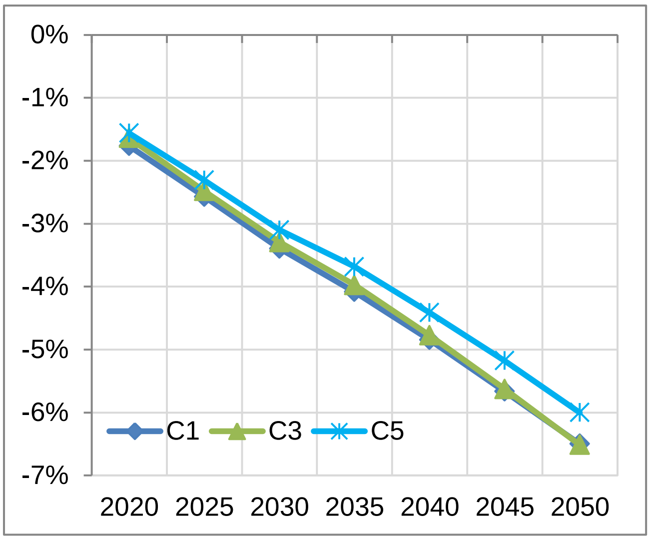
For the global economy, there is a decline of 2% in 2030 and 5% in 2050 in global GDP. In fact, the level of global reductions is similar in all linking scenarios, as in Figure C3d. Given the urgent need to transit towards a decarbonised global economy, this result suggests that if not combined with a meaningful emission reduction target, a market-based approach that put a price on emissions may fail to prevent further environmental effects.

Additionally, it demonstrates that the linkage architecture encourages change in behaviour so as to make decisions for internalising the cost of emitting on the basis of forthcoming net benefits, as international transfers. This is the case for China in all cases. For Brazil, linking to Europe produces the same stimulus. More importantly, it shows that the small size of the modelled Brazilian ETS matters, in the sense that it tends to benefit the most from cost savings mainly because it adopts the Industrial Plan mitigation target. Hence, whilst large markets seem attractive to Brazil, the Brazilian ETS is not such a tempting prospect for larger potential trading partners. As a result, such linkage provides minute contribution to addressing climate change in contrast to implementing a domestic sectoral ETS. One alternative is to regulate larger emitters in Brazil, for example, those associated to land use and deforestation. On the partner’s perspective, it calls for wider participation and more stringent limits on emissions levels.

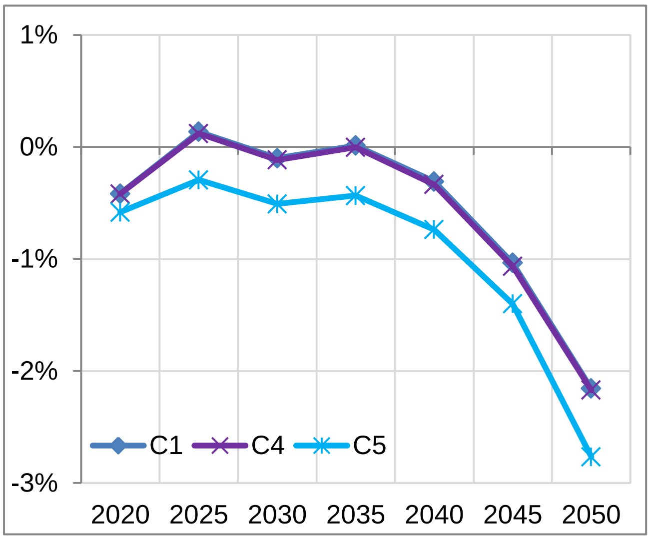
Figure C2 – Changes in GDP of partners compared to No-Policy Scenario



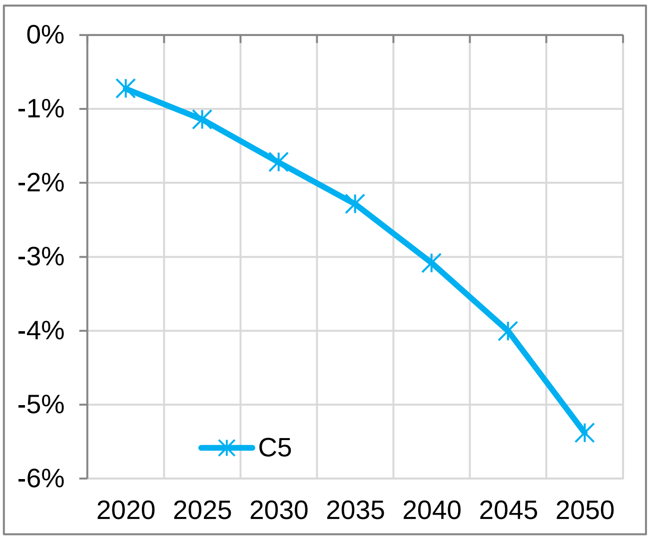
(a) Europe



(b) Latin America

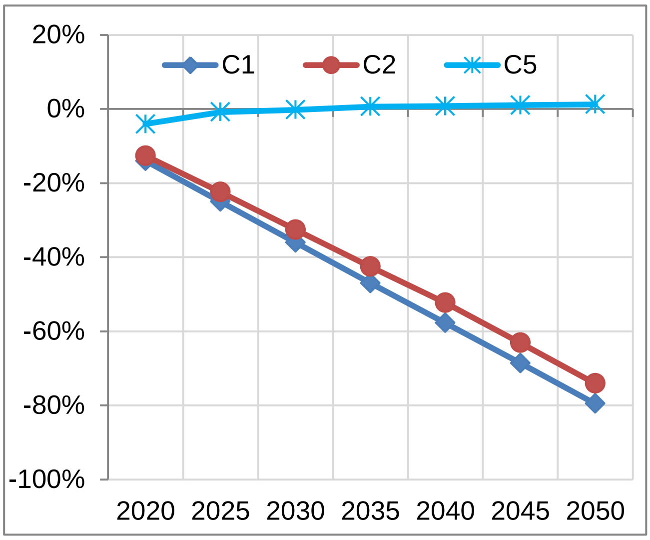


(c) China

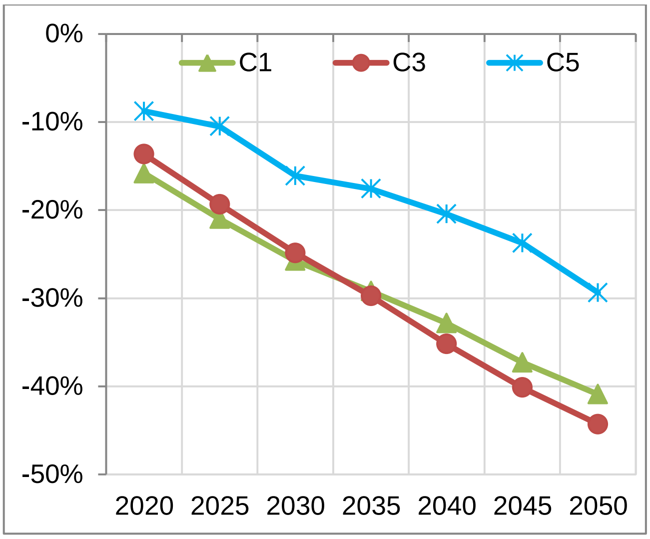


(d) World

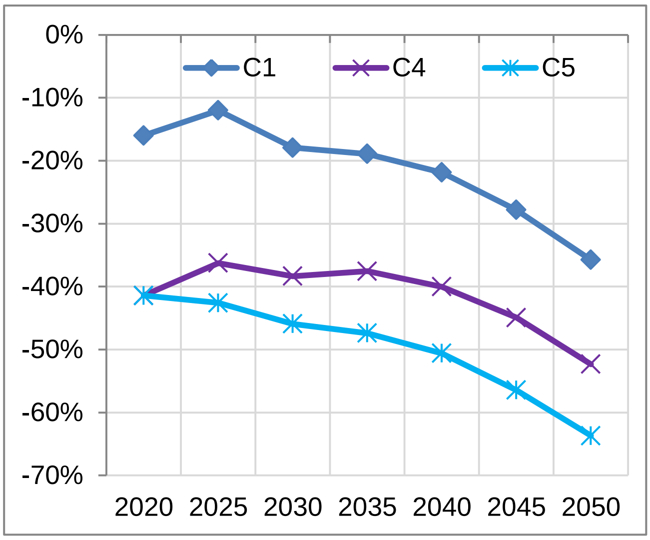
Figure C 3 – Changes in ETS emissions of partners compared to No-Policy scenario



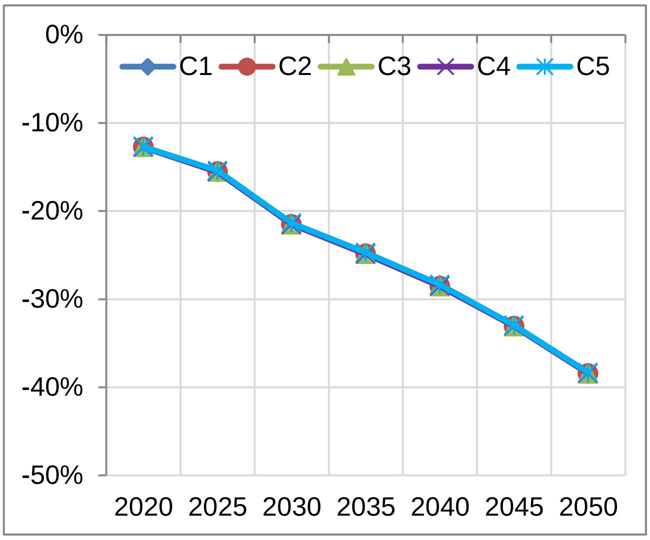
(a) Europe



(b) Latin America



(c) China



(d) World

APPENDIX D – A GLOBAL COOPERATION VIA SECTORAL ETS

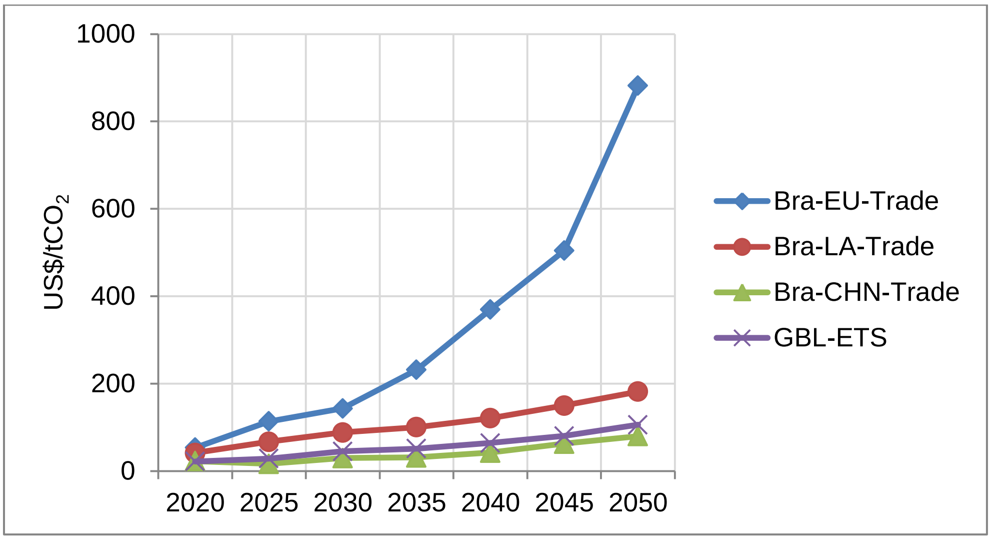
The literature usually points out a global market as the most cost-effective manner to tackle climate change. The global carbon price could ensure participation of all regions thereby preventing free riding and carbon leakage. Despite the relative progress on international climate negotiations, including through the introduction of provisions for linkages, such a global cooperation has so far not been agreed. In the future, however, a global international cooperation could be an option, compared to bilateral or multilateral cooperation agreements. To put this in perspective with the proposed bilateral linkages, the global sectoral ETS is analysed in order to verify whether it is compatible with a deep abatement level and at what cost to Brazil and the other trading partners. Scenarios are described in Table D1 below.

Table D1 – Summary of scenarios

Scenarios	Description
Bra-EU-Trade	a Bra-EU link, no banking, no revenue recycling
Bra-LA-Trade	a Bra-LA link, no banking, no revenue recycling
Bra-CHN-Trade	a Bra-CHN link, no banking, no revenue recycling
GBL-ETS	a sectoral Global ETS, no banking, no revenue recycling

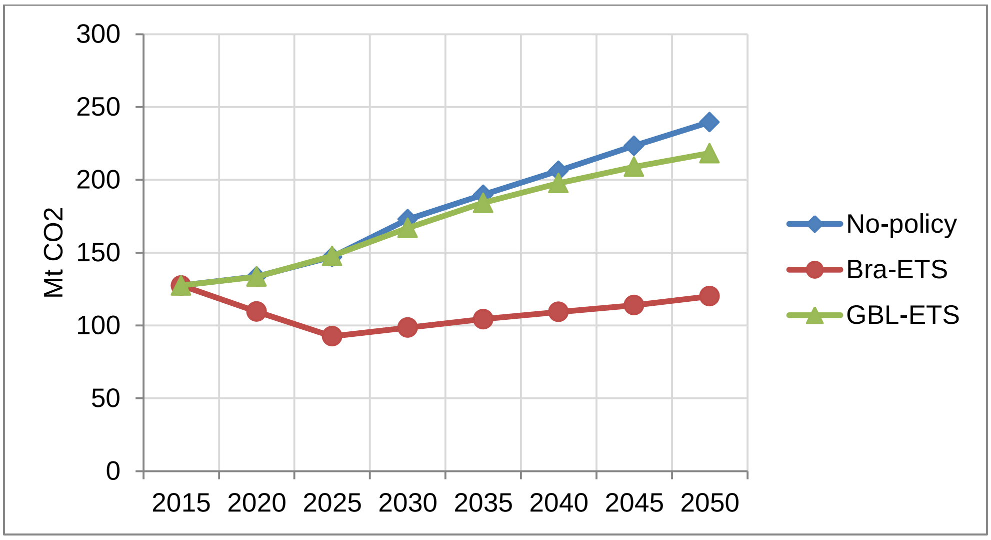
The carbon price for 2020-2050 is depicted in Figure D1. Among the proposed linkages, linking a developed-developing region implies the greatest costs. This is due to the admitted level of mitigation for both Brazil and Europe. Under the modelled framework, global sectoral trading represents a less costly option for Brazil, with a carbon cost of US$45/t$CO_2$ in 2030 and US$106/t$CO_2$ in 2050. For electricity and energy-intensive industries in Brazil, the cost of releasing one tonne of CO_2 in the atmosphere within an isolated sectoral ETS is US$202 and US$305 in 2030 and 2050, respectively.

Figure D1– CO_2 prices Bilateral vs. Global linkage



Participating in a global ETS has the advantage of being a larger market, with a high degree of heterogeneity and abatement opportunities across regulated entities. Since abatement occurs where it is cheap to do so, regulated sectors benefit by the flexibility of choosing the level of production and corresponding level of emissions, as it is possible to obtain permits in the market. If compared to a bilateral linkage with Latin America, the costs of a sectoral ETS are reduced for Brazil.

Figure D2– ETS emissions for Brazil Domestic vs. Global sectoral ETS



On the other hand, a bilateral link with China is the most cost-effective policy, but it produces the reversal incentive on emissions, that is, it increases sectoral emissions in Brazil. The effect of a global ETS on sectoral emissions is similar given that emission reduction remains unchanged until 2030 relative to No-Policy and from 2035 slightly decrease, corresponding to 22 million tonnes less than No-Policy in 2050. In the same period, the Brazilian ETS curbs 50% of No-Policy emissions, that is, 120 million tonnes of carbon.

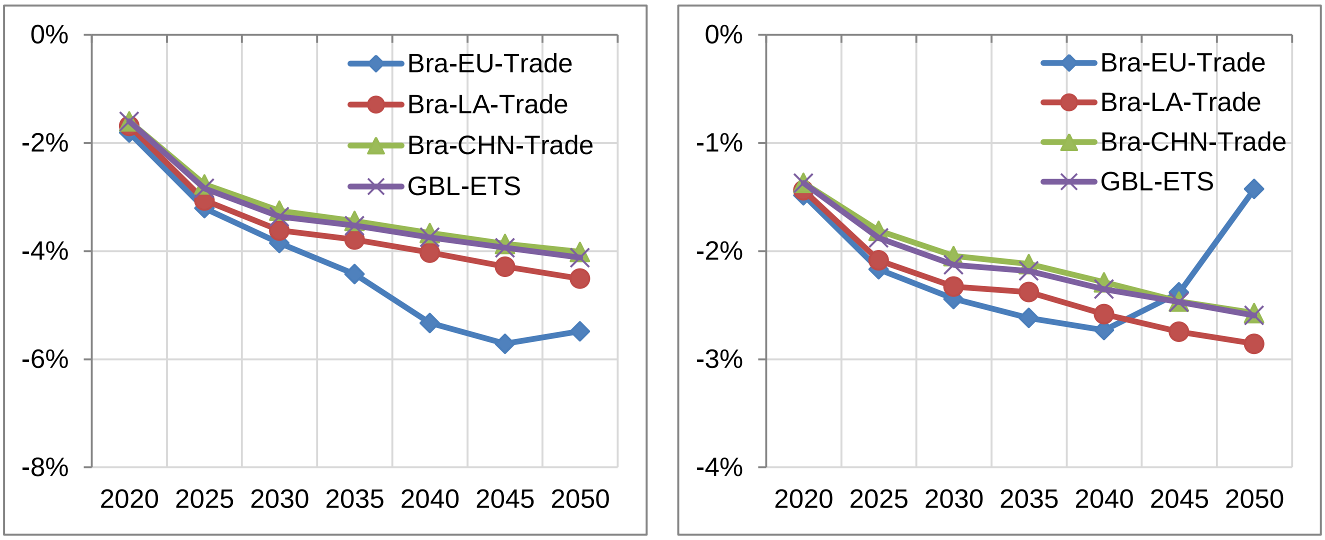
Within a global ETS, there is an income flow from Brazil to other regions, since it exhibits an importer of permits profile. Similarly, it is more cost-effective for Europe and Latin America to acquire permits instead of further mitigating. One of the greatest suppliers of permits in such a worldwide system is China, which receives approximately US$40 billion in 2030 and US$170 billion in 2050. Note that in this scenario most of those permits are bought by Europe, indicating that the target imposed is economically very limiting. A global linkage is a more strategic option for Europe than linking with Brazil. Latin America also has an import-led pattern, where total financial transfers correspond to similar amounts as the Brazilian ETS. This results from the homogeneous macroeconomic and energy profile among them, which determines the size and relevance of the covered sectors, as well as the opportunities of abatement available.

Table D2 – Total financial transfers of CO_2 permits within the Global sectoral ETS (in 2007 US$ billion)

Region	Bra	EU	LA	CHN
2020	0.5	3.7	0.8	-0.3
2025	1.6	12.1	1.8	-20.1
2030	3.1	29.1	2.9	-38.9
2035	4.1	45.2	4.5	-60.6
2040	5.7	71.1	6.5	-88.1
2045	7.6	107.4	9.7	-122.9
2050	10.4	167.3	11.7	-168.4

Macroeconomic effects on Brazil are portrayed in Figure D3 whereas those for considered partners are presented in Figure D4. For Brazil, GDP losses are the lowest if integrating a global ETS, in contrast to linking with Europe that leads to a long-term reduction of 6% compared to the No-Policy scenario.

Figure D3 – Changes in GDP (a) and Welfare (b) for Brazil: Bilateral vs. Global linkages



(a) GDP (b) Welfare

Other potential partners are affected differently. In the short term, Latin America faces the greatest drop in GDP at 3.2%, and only approximately 0.5% of GDP reduces from the No-Policy scenario of Europe and China. To the extent with which the targets get stricter, there are negative effects on Europe's GDP of 7.4%, a 6% decrease in Latin America and 2.8% in China. Specifically for Latin America, failure to foster economic development and poverty reduction is detrimental and could be perceived as politically unacceptable for the region. Despite the downward trend of changes in welfare, a global ETS has lower effects in Brazil than linking to Latin America and it is identical to joining the Chinese ETS. In fact, if in a global ETS, the welfare of Latin America declines 2.3% in 2030 and 16.2% of sectoral emissions are reduced. Further mitigation is achieved in Latin America even if in a global market, which has an additional impact on welfare in 2050. For China, reducing 50% of sectoral emissions in the global market implies an average of 0.2% gain in welfare in the period.

Table D3 – Changes in GDP, welfare and Sectoral emissions from the No-Policy scenario in 2030 and 2050 for considered trading partners

	Region	GDP	Welfare	Sectoral Emissions
2030	Europe	-0.4%	-1.4%	-0.3%
	Latin America	-3.1%	-2.3%	-16.2%
	China	-0.5%	0.3%	-27.3%
2050	Europe	-7.4%	-7.4%	1.2%
	Latin America	-6.0%	-4.8%	-29.4%
	China	-2.8%	0.2%	-49.8%

A global international cooperation via sectoral ETS is, to a certain degree, relevant to help bring all regions together into limiting the worst impacts of climate change. The findings could not measure whether or not this level of abatement is consistent with avoiding a 2°C rise in global temperatures. However, it demonstrates that putting a price on emissions globally has the potential to bring about emissions reductions whereas trading permits reduces economic costs of capping emissions.

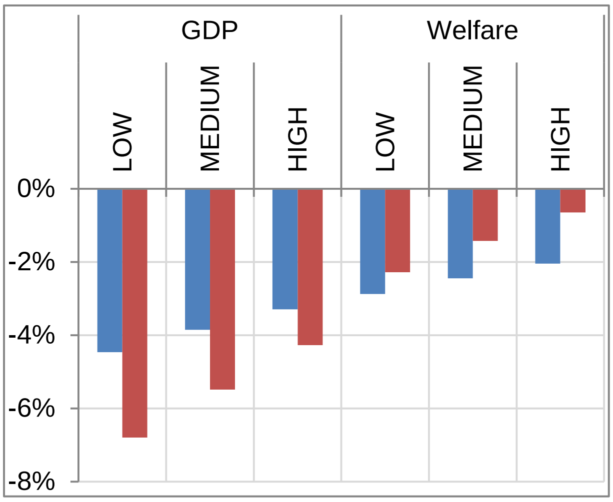
Ultimately, it indicates that developing countries would be encouraged to participate in the global system only if it is in accordance with the principle of common but differentiated responsibilities, as well as their respective capabilities. In this case, it is recommended to impose more stringent targets on larger emitters in order to align with historic evidence. By taking deeper reduction targets, efficiencies from trading emerge and do not compromise sustainable development in developing countries, or the efforts to prevent dangerous global climate change. To accomplish this long-term goal, supplementary policies are also needed.

APPENDIX E – SENSITIVITY ANALYSIS

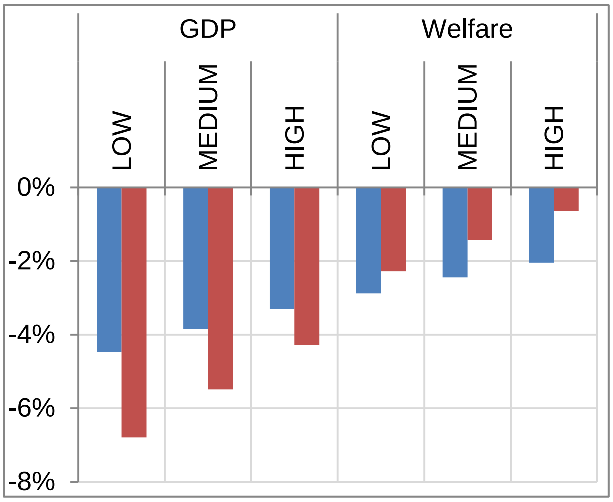
As a robustness check of the standard linkage scenarios (Bra-EU-Trade, Bra-LA-Trade and Bra-CHN-Trade), this appendix includes alternative policy simulations. Whilst the first sensitivity analysis concerns the GDP assumptions, the second is related to improvements in energy efficiency. It is considered an upper and lower rate for each variable, as described in Chapter 3. Given the internal economic structure of the linkage participants, uncertainties about long-term GDP and energy efficiency may play an important role in determining the implications of an emissions trading policy on low-carbon economic development. For the Brazil-Europe linkage, results for three selected endogenous variables, namely GDP, welfare and carbon prices, are depicted in Figure E1 and Figure E2.

Qualitatively, results hold throughout all cases. In the Brazilian case, there are small differences in the magnitude of GDP variations, meaning that results obtained are sensitive to this parameter. Further energy efficiency or higher growth rates in either 2030 or 2050 reduces the climate policy costs. For Europe, results for energy efficiency are robust where the gap is minimal between the scenarios. Similarly to Brazil, the lower GDP growth is expected to be, the deeper adverse economic effects of a sectoral ETS in Europe. Even though carbon prices are hardly sensitive to the implemented changes in annual GDP growth rates, a slight decline to the extent which growth rates increase, is observed.

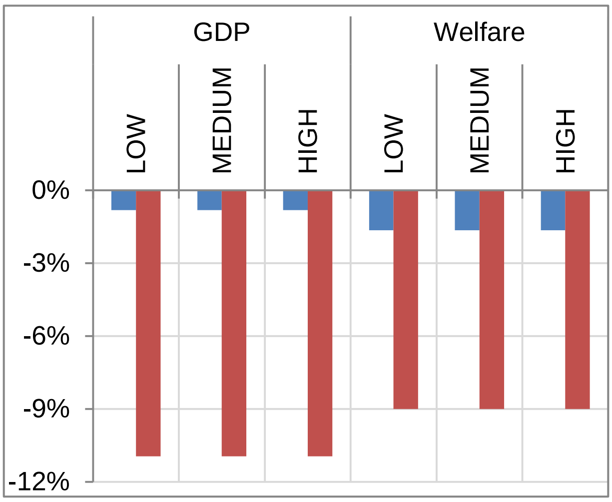
Figure E1 – Results for the Bra-EU-Trade scenario (% changes from No-Policy) under different (a) GDP growth and (b) energy efficiency assumptions for Brazil and Europe in 2030 and 2050



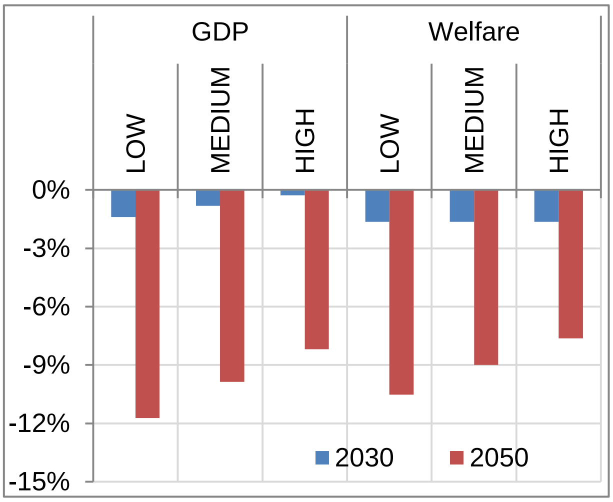
(a) Brazil – GDP growth



(b) Brazil - Energy Efficiency

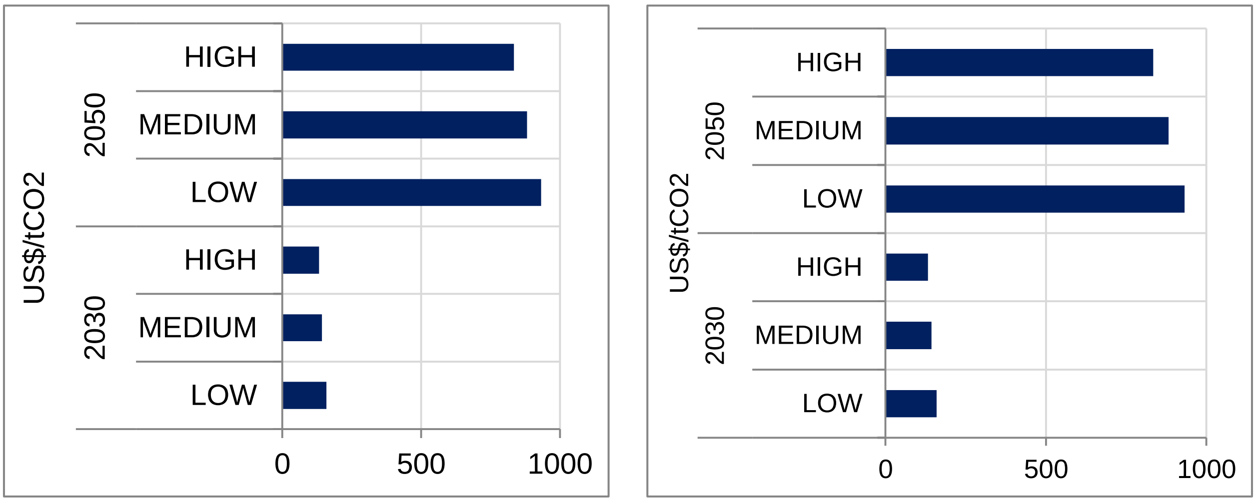


(c) Europe – GDP Growth



(d) Europe - Energy Efficiency

Figure E2 – Results for the Bra-EU-Trade scenario (% changes from No-Policy) under different growth (a) and (b) energy efficiency assumptions for carbon price in 2030 and 2050

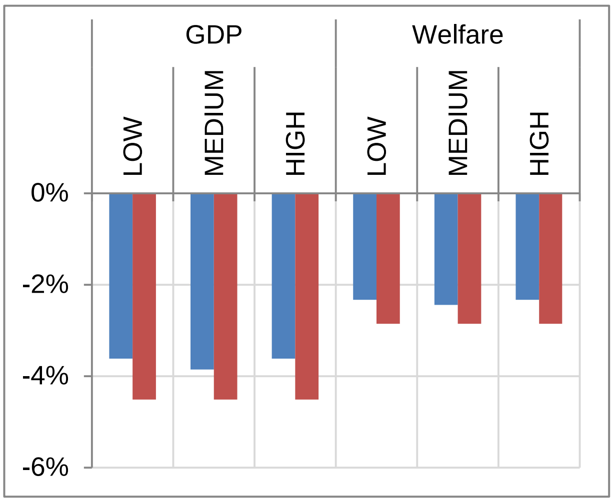


(a) GDP growth (b) Energy Efficiency

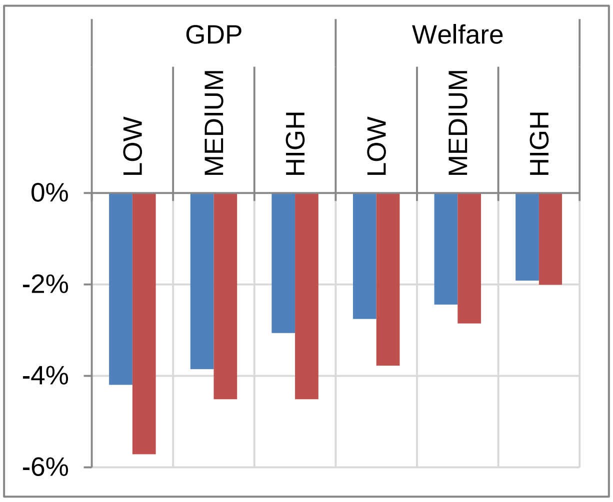
Mitigation strategies differ across regions and depend on both the level of economic development and growth perspectives, but for developing countries, energy efficiency plays an additional role in helping to drive emission reductions. In Figure E3a and Figure E3c, GDP and welfare for Brazil and Latin America under the linkage hardly react to variations in growth rates assumptions. On the other hand, those variables are more sensitive to upper and lower changes in energy efficiency, as evidenced in Figure 36b and Figure 36d. Specifically, long-term reactions of GDP are more pronounced than welfare's so that the gap between GDP scenarios is 1.3%.

In this sense, economic impacts of sectoral ETS linkage between Brazil and Latin America largely rely on energy efficiency assumptions. Similarly, the carbon prices respond differently to this parameter either in 2030 or 2050. Hence, the availability and cost of low-carbon technologies have a strong impact on the simulated carbon price level by influencing energy efficiency improvements. The higher they are, the lower the carbon costs as a result of market forces.

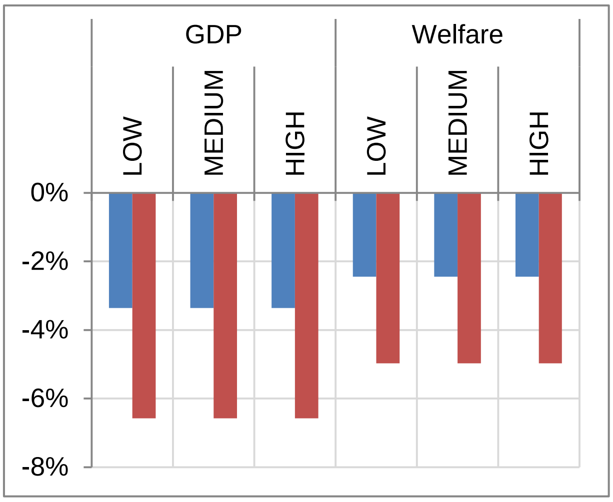
Figure E3 – Results for the Bra-LA-Trade scenario (% changes from No-Policy) under different (a) GDP growth and (b) energy efficiency assumptions for Brazil and Latin America in 2030 and 2050



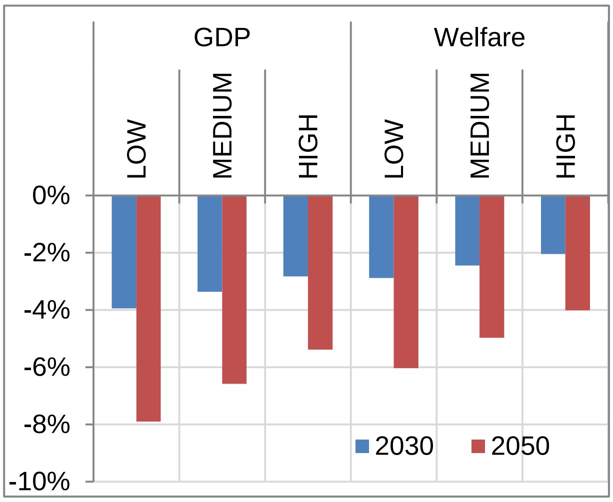
(a) Brazil – GDP Growth



(b) Brazil – Energy Efficiency

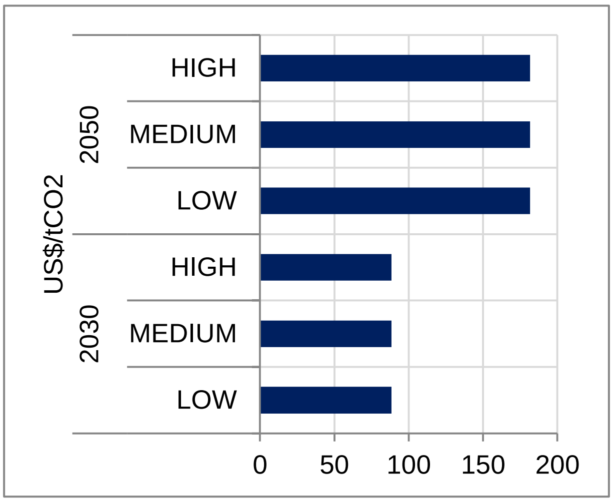


(c) Latin America – GDP growth

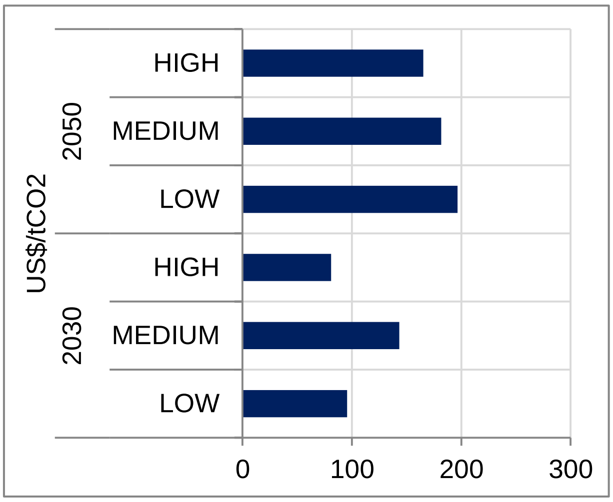


(d) Latin America – Energy Efficiency

Figure E4 – Results for the Bra-LA-Trade scenario (% changes from No-Policy) under different growth (a) and energy efficiency (b) assumptions for carbon price in 2030 and 2050



(a) GDP growth

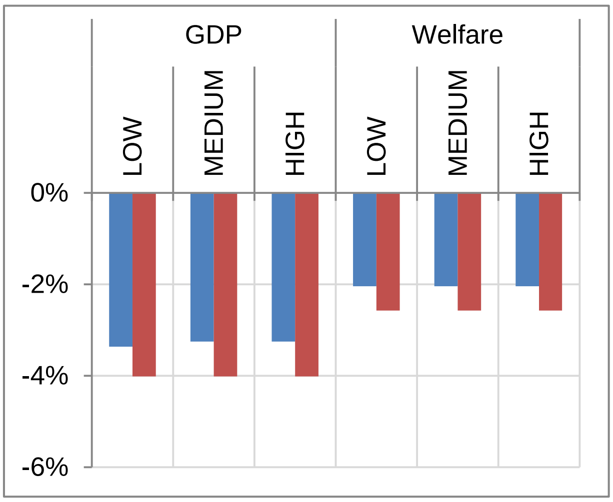


(b) Energy Efficiency

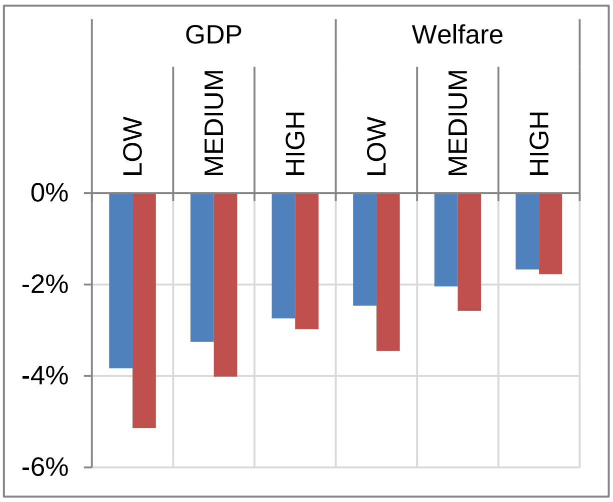
A sensitivity analysis of GDP growth rate and energy efficiency is provided for the Brazil-China linkage in Figure E5 and Figure E6. In terms of GDP growth assumptions, results hold throughout all cases for both Brazil and China and demonstrate that GDP and welfare analysis previously presented are robust to the sensitivity test. On the other hand, economic impacts of sectoral ETS linkage differ along with variations of energy efficiency levels.

Whilst endogenous macroeconomic variables of Brazil are negatively affected across the period of compliance in Figure 49b, those in China react positively to improvements in energy efficiency. In the long-term, the costs of imposing a carbon constraint are significant for both regions, but China is more sensitive to this parameter, reaching 4.7% and 2.8% decrease in GDP and welfare, respectively. The magnitude of this effect is similar to Brazil's, where GDP reduction totalises 5.1% while welfare reduces 3.5%. Results for the carbon price in Figure E6 also indicate a different response to energy efficiency levels in contrast to GDP. A higher degree of energy efficiency implies a reduced cost of carbon.

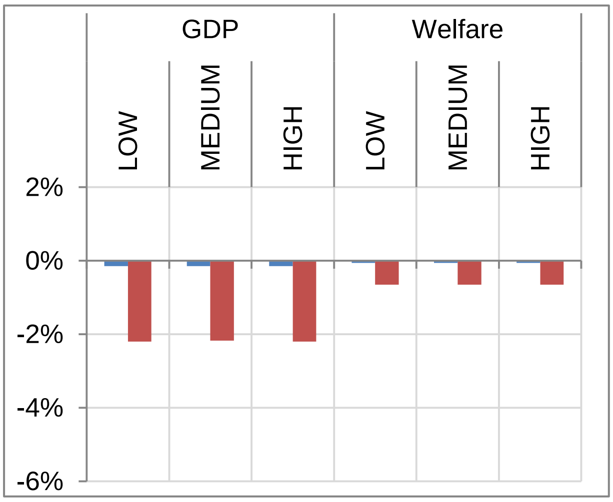
Figure E5 – Results for the Bra-CHN-Trade scenario (% changes from No-Policy) under different (a) GDP growth and (b) energy efficiency assumptions for Brazil and China in 2030 and 2050



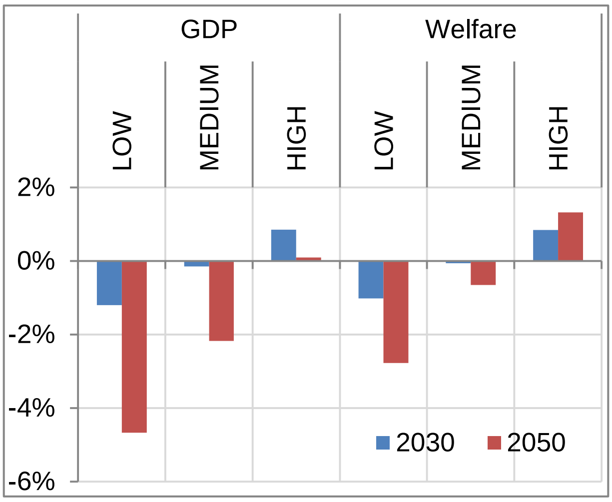
(a) Brazil – GDP Growth



(b) Brazil – Energy Efficiency

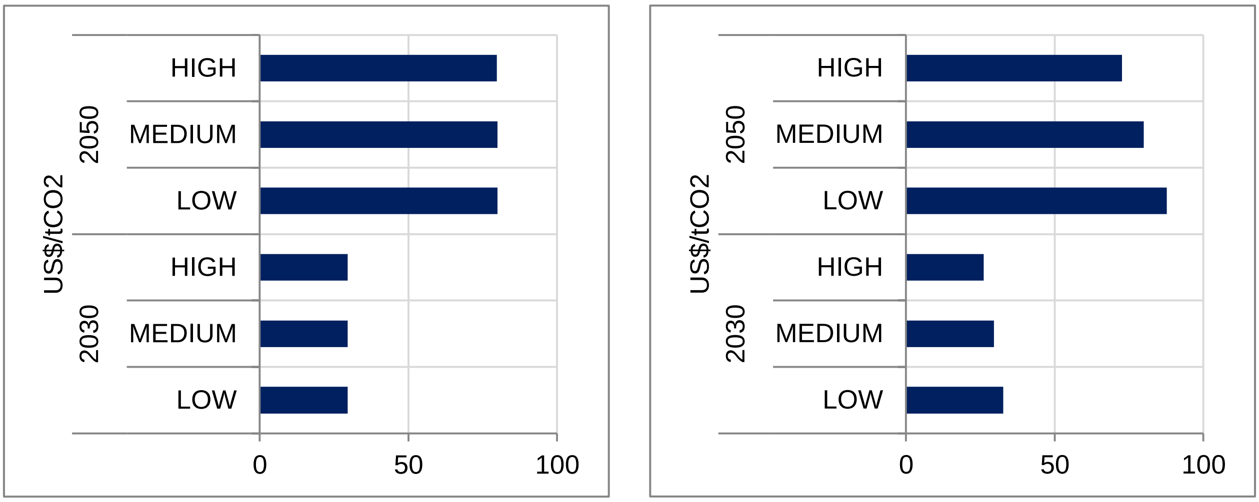


(c) China – GDP growth



(d) China – Energy Efficiency

Figure E6– Results for the Bra-CHN-Trade scenario (% changes from No-Policy) under different (a) GDP growth and (b) energy efficiency assumptions for carbon price in 2030 and 2050



(a) GDP growth

(b) Energy Efficiency