Implementation of an emission trading scheme in Japan: Some food for thought

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October 2011

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Acknowledgement

The author is grateful and indebted to Xiangchun Lu (Tohoku University) and Jusen Asuka (Tohoku University and IGES).

Publisher

Publisher i.e. Climate Strategies 2011

For citation and reprints, please contact the publisher Climate Strategies

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Executive Summary

In its "Action Plan for Achieving a Low-carbon Society", Japan has committed itself to the long-term goal of "halving total global greenhouse gas (GHG) emissions by 2050 from its current level of emissions". Under the 2009 Copenhagen Accord, the government made the commitment to reduce the Japanese GHG emissions by 25% between 1990 and 2020.[1] This requires increased efforts to reduce greenhouse gas emissions from all Japanese sectors. All the more so as in 2008 a review of the Japanese climate policy estimated that the country would face excess emissions of 22-36 MtCO2e by 2012 compared to the commitment under the Kyoto Protocol.

After several years with a voluntary emissions trading system (ETS), a consensus amongst Japanese industry has emerged to implement a mandatory scheme. A smooth transition period is planned, keeping the current voluntary ETS before making it obligatory. During this period public authorities have to define the ETS structure, especially the scheme period, the total quantity of emission allowances, the entities covered and the allocation method (Ministry of the Environment, 2009). Particular attention is given to the scheme's impact on the international competitiveness of Japanese firms as well as on the risk of carbon leakage. This risk refers to impacts from carbon pricing on the operation and investment decision of manufacturers. Trade exposed, energy-intensive industries could replace domestic production by imports, or could relocate production to foreign countries in order to circumvent carbon costs. This will, however, undermine the environmental effectiveness of the ETS.

The aim of this paper is to quantitatively assess different design options for an ETS in Japan, especially their impact on carbon leakage as well as on industrial production. We develop a model, called CASE-ASIA, which represents several Japanese industrial sectors which would be strongly impacted by implementing a CO_2 constraint. With this model, we analyse different design options of a mandatory ETS as well as different "anti-leakage" options. Then, we provide a quantitative assessment how these different schemes impact on the activity of the different industries included in the model. The results show that there is not a "best option" and that the Japanese government should decide where its priorities lie. The analysis makes explicit the trade-offs implied by the different schemes and provides "food for thought" for the policy-makers.

This paper is organised as follows: Section 1 briefly introduces some elements of context; Section 2 presents the model CASE-ASIA and Section 3 the different simulations made; Section 4 discusses the results.

[1] The Fukishima nuclear accident might compromise these decisions about both the global emission reduction target and the start date of the ETS.

1 The context

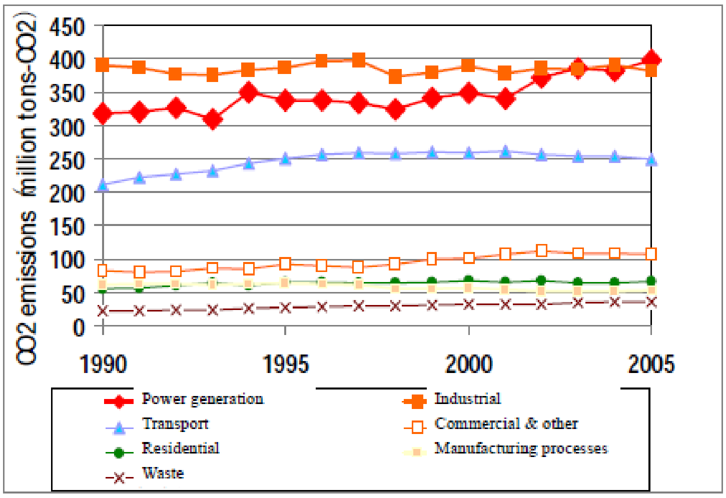
1.1 Development of GHG emissions in Japan

Under the Kyoto Protocol Japan has made a commitment to a 6% reduction in greenhouse gas emissions compared to 1990. According to Kiko Network (2008), Japan's GHG emissions amounted to 1.34 billion tons CO_2e in 2006, representing a 6.2% increase compared to 1990, and representing a 12.2% gap compared to the reduction target of 6%.

Since 1990, there has been an increase in CO_2 emissions in transport, the commercial and the residential sectors. The emissions from the industrial sector (manufacturing) have decreased by 4.6% since 1990, but this sector continues to account for 36% of emissions (see Figure 1).

Figure 1 also shows that power generation is the sector with the highest emission increases since 1990. According to Kiko Network (2008), more than half of the increase in Japan's emissions from 1990 to 2006 was due to power plants. This increase is explained by a drop in electricity consumption in the commercial and residential sectors and by the increase in emission factors in the electricity sector. Indeed there has been an increasing use of coal-fired power plants, which has led to an increase in emissions from coal-fired power plants greater than the increase in Japanese overall emissions since 1990.

Figure 1: Sector-by-sector trends in CO_2 emissions (direct emissions)



Source: Kiko Network (2008)

1.2 From a voluntary to a mandatory emission trading scheme for industry

The Japanese climate policy has been strengthened through a step-by-step approach, rather than via a once and for all introduction of policies and measures (Kimura and Tuerk, 2008). Japan started with the Keidanren Voluntary Action Plan, based on the voluntary adoption of intensity targets, and then introduced a voluntary emission trading scheme (ETS). Keidanren's Voluntary Action Plan was developed in 1997 with the aim of stabilizing energy-source and industrial-source CO_2 emissions at their 1990 level by 2010. It has been reviewed and strengthened by the government in the Kyoto Target Achievement Plan (KTAP), while being not legally binding.

In 2005 Japan introduced a voluntary emission trading scheme, the JVETS, based on absolute targets. This scheme however attracted only a very small number of participants. For participating firms, one-third of the cost of new facilities to reduce emissions was borne by the government (Jones and Yoo, 2009). Firms that failed to achieve their objective had to purchase credits from firms that had achieved larger-than-targeted reductions or return the subsidy to the government. The participating firms accounted for less than 1% of CO_2 emissions from the industrial sector, which provided an incentive for the government to change its policy (Kimura and Tuerk, 2008).

In 2008, a consensus to implement a mandatory ETS in the Japanese climate policy has emerged following a review of the KTAP which estimated that Japan will face a shortage 22-36 $MtCO_2e$ by 2012.[2] However, a

[2] See Kimura and Tuerk (2008).

transition period to a mandatory ETS is foreseen and it is expected to allow for a smooth transition from the current approach under Keidanren's Voluntary Action Plan. In October 2008, the government launched an experimental emissions trading system that includes the JVETS as an option. The main objective is to gain experience in using an ETS system. After an intensive recruitment process, 715 firms agreed to participate, including 521 firms which agreed to targets accounting for about two-thirds of CO_2 emissions by the industrial sector.

Moreover, an Advisory Committee on the Emissions Trading Scheme has been established by the Ministry of the Environment. Its objectives are, among others, to make an in-depth study on the specific design of a possible ETS, in particular its effectiveness and feasibility and its impacts on domestic industries and employment. Moreover, the Committee must evaluate possible options for the industrial sectors that would be substantially affected in terms of international competitiveness or would run a major risk of carbon leakage.

1.3 The anti-leakage policy options in an ETS

When designing an ETS, two main options to reduce carbon leakage are generally considered: free allocation and border adjustment. A border adjustment (BA) is a trade measure designed to level the playing field between domestic producers facing climate policy measures and foreign producers with no or little constraint on their GHG emissions. They could be designed for example as import tariff, export rebate or export tariff. As a trade measure, a BA may be contested by a World Trade Organisation (WTO) member under its dispute settlement mechanism. There is a considerable literature debating the legality of BA for climate policies under WTO rules. Recent analyses, including a report by WTO/UNEP (2009), concluded that, under specific conditions, such a measure may be WTO compatible.[3] A BA was the option retained in the American Waxman-Markey bill which suggested to implement a cap-and-trade scheme from 2016 onwards with an "international reserve allowance program" for imports from 2020 (James, 2009; van Asselt and Brewer, 2010).[4]

In the EU, the Directive 2009/29/EC, which revises the EU ETS for Phase III (2013-2020), lists the use of a BA on imports but the EU finally opted for continued free allowance allocation to the "sectors or subsectors which are exposed to a significant risk of carbon leakage" (Article 10a-12). As explained in Matthes and Monjon (2008) and Ellerman (2008), since allocation in the EU ETS is linked to production capacity, not to actual output, this can reduce the incentive to relocate production capacity (investment leakage). However, it cannot reduce the incentive to use existing plants abroad rather than EU plants (operational leakage). A suggested approach to tackle this problem is output-based allocation (OBA), under which for example a steel producer would receive a given amount of allowances per tonne of steel actually produced. Practically, this requires an update of the allocation when production is known. Output-based allocation thus has advantages and disadvantages which are discussed in Quirion (2009). In particular, an output-based allocation, called "home rebate" by Fischer and Fox (2009a), levels the playing field between the domestic and the foreign market. However, this does not give an incentive to reduce emissions by reducing consumption.

1.4 Sectors exposed to the risk of carbon leakage under a Japanese ETS

In order to identify the sectors impacted most by a carbon cost in Japan, Asuka et al. (2009) followed the method proposed in Hourcade et al. (2007). They evaluated the carbon intensity or the "value at stake" of the Japanese (sub-) sectors defined as:

$$\text{Carbon Intensity} = \frac{\text{Purchasing Costs of Emission Allowances}}{\text{Gross Value Added (GVA)}}$$

The numerator shows the range of carbon costs expressed in terms of costs borne by the corporations to purchase emission allowances for their direct and/or indirect CO_2 emissions. The denominator shows the extent of industry sectors' profits – expressed in gross value added. To estimate the cost of the indirect emissions, it is assumed that 100% of the additional costs from electricity production will be passed on by raising electricity prices.

The Net Value Added at Stake (NVAS) for each industry is calculated by assuming that the Government of Japan auctions allowances to the power sector only and gives free allowances to the other sectors. Consequently, the numerator only takes into account the increase of the electricity price. The Maximum

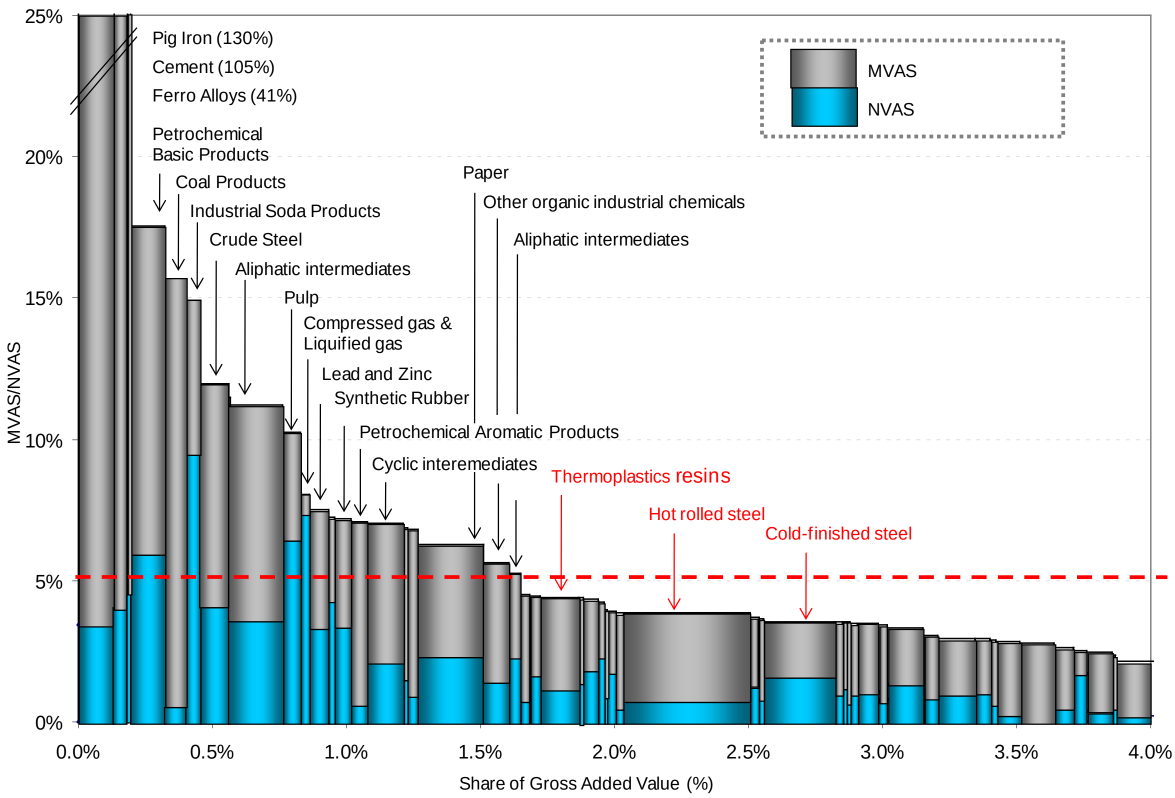
[3] See Droege (2011) on this issue.

[4] The bill was approved by the House of Representatives on June 26, 2009 by a vote of 219-212, but was rejected by the Senate.

Value Added at Stake (MVAS) is calculated by assuming the auctioning of the allowances to all industry sectors. The allowances price is exogenously set at 3000Yen/tCO_2e.

Figure 2 shows the calculation results of carbon intensity for the sectors with a MVAS of 2% or greater. The vertical axis indicates MVAS and NVAS, with the horizontal axis showing the share of each sector in overall domestic production. Only sectors with a MVAS of 5% or greater are shown with names. The analysis covers 401 sectors designated under the inter-industry relations table from the year 2000.

Figure 2: Sector-by-sector evaluation of the impact of a carbon constraint

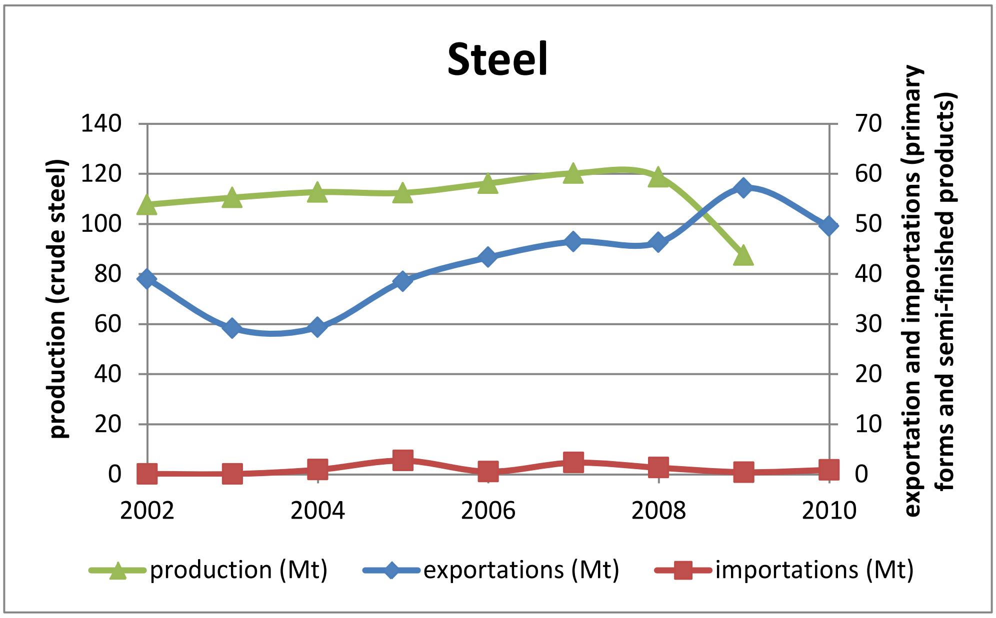


Source: Asuka, Kanamoto and Xiang Chun (2009)

Asuka et al. (2009) conclude that the share of industry sectors with a MVAS of 2% or greater represents 3.2% of total Japanese GDP. There are 17 industrial (sub-) sectors with a MVAS of 5% or greater. Lastly, pig iron and cement sectors have a relatively higher MVAS compared to the other sectors studied.

Following the conclusion of Asuka et al. (2009), our analysis focuses on some sectors exposed to carbon leakage, in particular on the steel and cement sectors, as well as the petrochemical basic products. The first two sectors are large net exporters (Figures 3 and 4), while the third sector is a net importer.

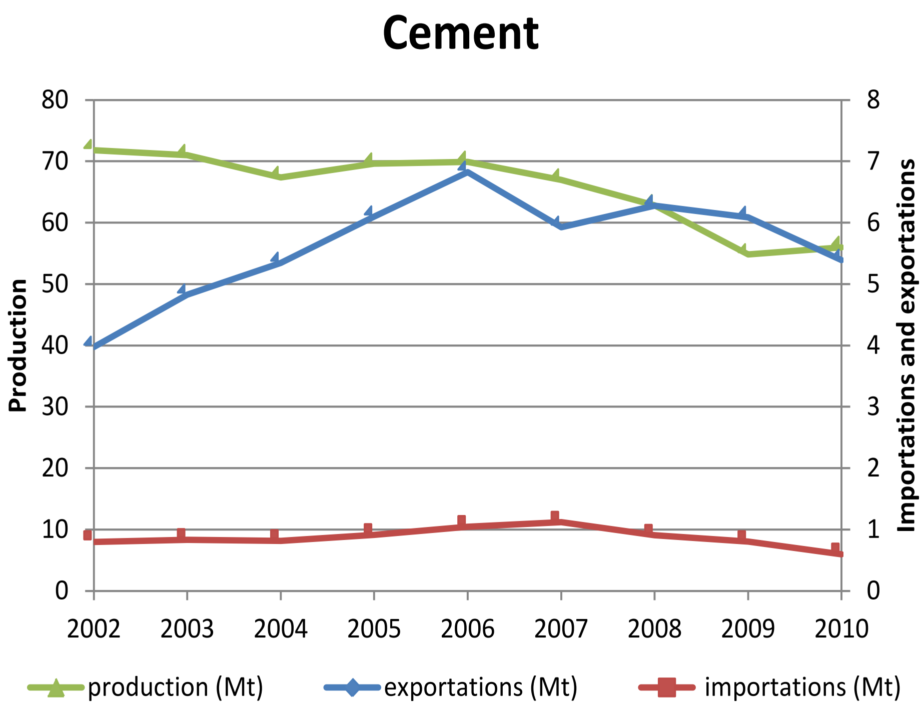
Figure 3: Japanese production, imports and exports of steel (in Mt)



Note: Codes: H2-7206 and 7207.

Source: UNCOMTRADE and IISI.

Figure 4: Japanese production, imports and exports of cement (in Mt)



Note: Codes H2-252321 and H2-252329.

Source: UNCOMTRADE and USGS.

2 The Model[5]

The model CASE-ASIA follows the logic of the CASE II model developed to represent the Emission Trading System (EU ETS) implemented in EU since 2005 (Monjon and Quirion, 2011a). CASE-ASIA is a static and partial equilibrium model which represents four sectors: cement, steel, refined petroleum products and electricity. The model comprises two regions: Japan and the Rest of the World (RoW). All sectors have a potentially large cost impact from carbon pricing but will face different direct and indirect emissions costs as well as different cost structures (Asuka et al., 2009).

The model aims to evaluate the impact of different designs for an ETS in 2020 with respect to: production levels, price levels and trade flows in each industry. The model also allows for the calculation of the leakage-to-(emissions) reduction ratio for each sector and for the whole ETS.

When carbon pricing policy is carried out in Japan, domestic firms face three types of additional costs:

- Abatement cost: The abatement costs are based on Marginal Abatement Cost Curves (MACC) taken from the POLES model for the year 2020 in which Japan is explicitly modelled. In POLES, the MACCs are available for CO_2 energy emissions from, among others, non-mineral materials, chemical, steel and electricity sectors. The MACCs have been used to define a curve which gives, for each CO2 price, the decrease in unitary emissions. The abatement costs are incorporated into the model as variable costs, i.e. per product ton.
- Purchase of allowances: Firms may need to buy allowances, which increases the variable costs, or sell the allowances they do not use, which decreases the variable costs.
- Indirect cost from electricity pricing: The marginal production cost of cement, refined petroleum products and steel firms increases when there is a rise in the electricity price. We assume a cost pass-through of 100% in the power sector, whatever the policy scenario modelled. This part of the cost increase corresponds to the numerator of the NVAS defined in Asuka et al. (2009).

2.1 Sector assumptions

According to Hourcade et al. (2007) the risk of carbon leakage is highest for carbon-intensive primary commodities and semi-finished products. Consequently, the CASE-ASIA model focuses on this stage of the production chain. For the steel sector, we model semi-finished products (e.g. slabs), because they feature higher CO_2/turnover and CO_2/value added ratios relative to finished products; hence carbon leakage is more likely to happen at this stage of the production process. We aggregate long and flat products and both production routes (basic oxygen furnace and electric arc furnace).

In the model, all sectors consume electricity. We do not take into account the fact that some industrial companies produce their own electricity. Moreover, we do not consider electricity savings due to the rise in power prices but we extrapolate the recent trends in the electricity consumption per ton of product.

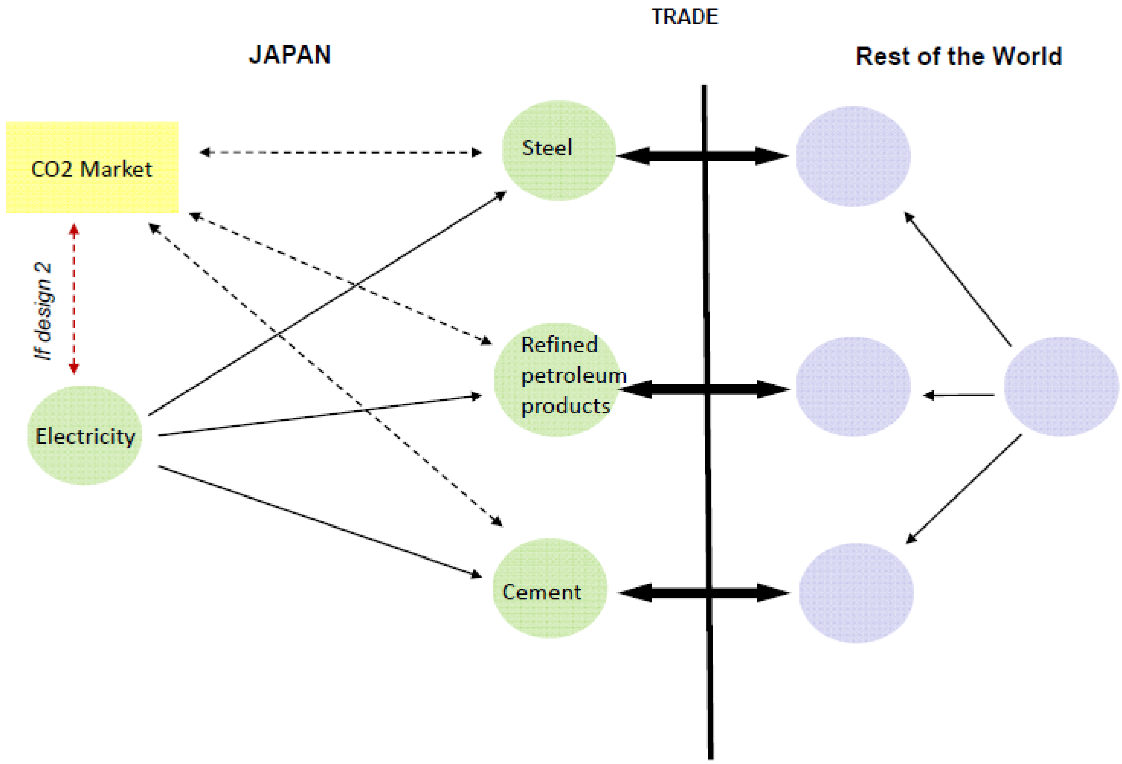
In the cement sector, we take into account the substitution of clinker (the CO_2-intensive intermediate product) with CO_2-free substitutes (e.g. fly ashes or blast furnace slag).[6]

2.2 Model Structure

All sectors included in the ETS are linked through the CO_2 market. The CO_2 price is determined by equalizing the demand and the supply of allowances: thanks to specific emissions abatement and the decrease in production, the sum of the emissions from these sectors equals the total amount of allowances allocated for free or auctioned.

[5] See Annex 1 as well.
[6] See Annex 1 for more details.

Figure 5: Model Structure



As explained below, we distinguish two designs depending on the inclusion of the power sector in the ETS. When the power sector is included in the ETS (Design 2), this sector intervenes on the CO_2 market as well. When the power sector is not in the ETS (Design 1), the red dotted line must be deleted.

We do not model emissions in the rest of the economy. These emissions could differ across our scenarios, due to some indirect effects (e.g. substitution between electricity with gas in building heating) but this effect is most likely to be negligible.

3 The Scenarios

Our goal is to evaluate the consequences of implementing a mandatory ETS in Japan. As explained above, neither the design of the ETS nor the allocation mode of the permits are decided yet. Based on the Executive Summary released by the Advisory Committee on the Emissions Trading Scheme (2008), several designs of the ETS and several allocation modes will be examined.

3.1 The Different Designs

The Japanese government envisages implementing an ETS not targeting the electricity sector, which might pay a CO_2-tax instead. This would be an important difference to the EU ETS setup. Following the Japanese policy plans, our scenarios will include two designs:

- **Design 1:** An ETS covering only the cement, steel and refined petroleum products. A CO_2 tax is applied to the electricity sector.
- **Design 2:** An ETS covering both the industrial and electricity sectors.

3.2 The Allocation Mode

The scenarios considered for the allocation mode focus on some representative configurations including specific treatment of the sectors exposed to carbon leakage risk (free allocation based on output, border adjustment). Each allocation mode is examined under both designs.

- **Auction** features full auctioning of allowances or a CO_2 tax in the electricity sector. In the other sectors, there is full auctioning of allowances, without rebating the auction revenues to the firms covered by the ETS, and without any anti-leakage provision.

- **BA_full** features **Auction** in the electricity sector. For the other sectors, there is 100% auctioning, with border adjustment on imports and exports and for direct emissions. The import and export adjustments are proportional to the Japanese average specific emissions (direct emissions).
- **BA_imports** features **Auction** in the electricity sector. In the other sectors, there is 100% auctioning with border adjustment only on the imports and for direct emissions. The import adjustment is proportional to the Japanese average specific emissions (direct emissions).
- **OBA** features **Auction** in the electricity sector. In the other sectors, there is free output-based allocation in exposed industries (cement, steel and refined petroleum products) for direct emissions. The amount auctioned is 83.3%[7] of the electricity sector emissions in 2005 when included in the ETS. In every other sector, the amount of allowances allocated per unit produced is calculated by applying a reduction ratio to the 2005 specific emissions. The reduction ratio is equal across sectors and calculated so that the emission cap is 83.3% of 2005 emissions.

Due to missing data on electricity price elasticity in the industrial sectors, we do not consider in this report an allocation mode which would attribute indirect emissions to the cement, steel and refined petroleum products sectors (downstream allocation). However, this allocation mode is envisaged in Japan and would need further investigation (Advisory Committee on the Emissions Trading Scheme, 2008).

We analyse eight climate policy scenarios and compare them to a no-policy scenario, which is simulated for 2020 without climate policy. The no-policy scenario is based on a growing Gross Domestic Product (GDP) and changing technical coefficients (specific emissions, specific electricity consumption). Other exogenous variables stay constant (in particular production costs).

3.3 Common features across the climate policy scenarios

The simulations will be made assuming unilateral carbon pricing by Japan. This ignores that the EU has already implemented an ETS covering GHG-intensive industries until 2020. However, as EU and Japan firms seldom compete on the same markets in cement, steel, and refined petroleum we stick to this assumption. Other efforts on carbon pricing in a number of regions, including the USA and South Korea, remain very uncertain and thus cannot be incorporated in the RoW part of the model. Moreover it is unclear how the national pledges made under the Copenhagen Accord and the Cancun Agreements would affect the production cost of these industries. Our focus is on comparing the relative impact of policy options rather than on estimating absolute values, and there is no reason to think that these uncertainties regarding international climate policies will change the ranking of policy scenarios in terms of their impact.

3.4 Emissions Reduction

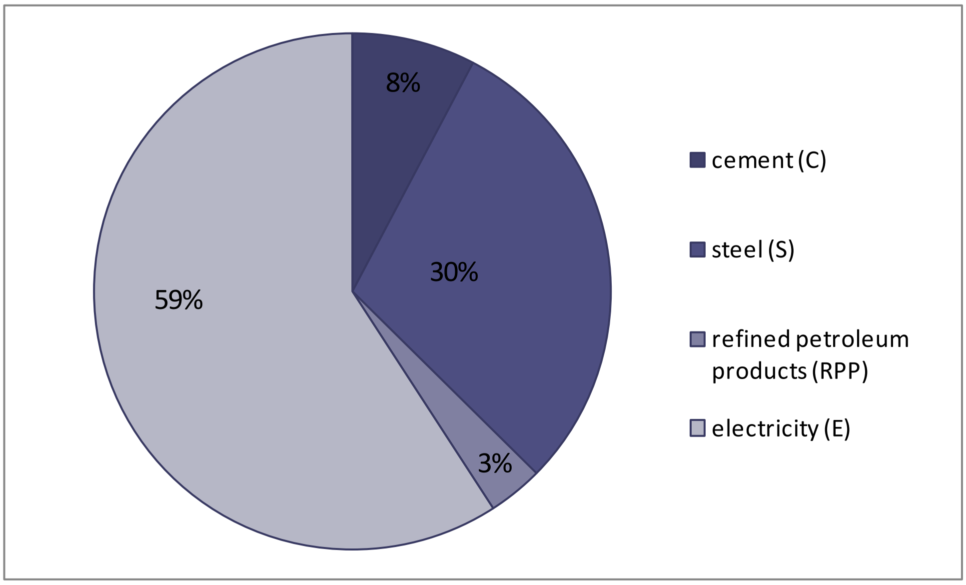
The national target announced by Japan under the future climate regime is to reduce its GHG emissions by 25% between 1990 and 2020, but there is not yet a decision about the reduction share to be delivered by sectors covered by the ETS.[8] Moreover, the 2011 Fukushima nuclear accident may modify the Japanese climate policy, both on global and industrial emission targets.

In 2005, the four sectors included in the model CASE-ASIA emitted around 615 MtCO2e. The electricity sector is the biggest emitter (59%) following by the steel sector (30%) (Figure 6).

[7] See the explanation in the following.

[8] A 25% reduction in emissions from 1990s levels by 2020 is equivalent to about a 37% reduction from 2005 levels (Pielke, 2009)

Figure 6: Share of the different sectors in 2005 GHG emissions



Based on the evaluation by the Japanese Ministry of Environment (2010), the Institute for Global Environmental Strategies (IGES) evaluated a specific reduction scenario between 2005 and 2020 for the following sectors:

Table 1: Emission reduction ratio between 2005 and 2020

Sector name	reduction ratio (2005-2020)
Cement	-7.5%
Hot rolled steel	-15.3%
Aliphatic intermediates	-5.6%
Electric power	-18.7%

Source: evaluation by IGES based on Ministry of Environment (2010)

It is then possible to apply these reduction ratios to the sectors included in the model and to evaluate the different caps applied in the ETS depending on the inclusion of the power sector. By applying the reduction ratio from table 1, the cap in 2020 would be 512.3 MtCO2e for the four sectors modelled, i.e. a decrease of 16.7%.

Table 2: the emission reduction between 2005 and 2020 depending on the design and of the sector

	Emission reduction 2005-2020	
	ETS (C, S and RPP)	Electricity sector
Design 1	-13%	-18.7%
Design 2	-16.7%	

Source: own calculations

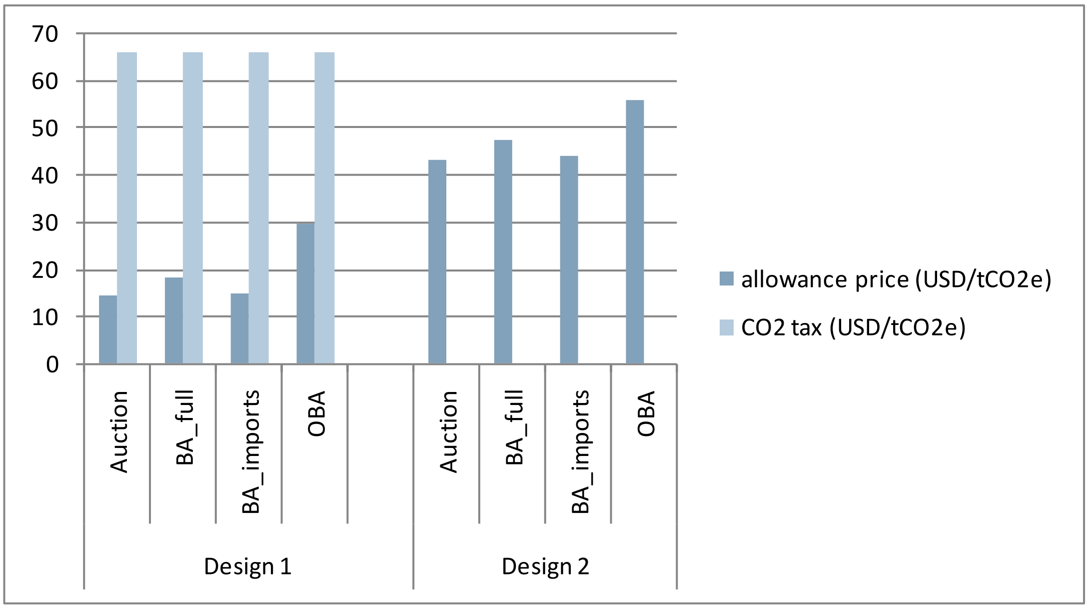
4 Results

4.1 The Design matters

The inclusion or the exclusion of the electricity sector in the ETS is a crucial element of the ETS approach modelled. When a CO_2-tax is applied in the electricity sector, its amount will need to be around USD 65-67/tCO_2 to reduce the emissions of the sector by around 18.7%, while the CO_2-allowance price will need to be between 14 and 30 USD/tCO_2e depending on the allocation mode (see Figure 7).[9]

Under Design 2, the allowance price is between 40 and 55 USD/tCO_2e, that is between the allowance price and the CO_2 tax under Design 1. Under Design 2, the allowance demand of the electricity sector pushes upward the allowance price. Consequently, more reductions in specific emissions are carried out in the industrial sectors than under Design 1, because these reductions are cheaper. The abatement costs between the sectors are equalised verifying the well-known equi-marginality principle.

Figure 7: Allowance price and CO2 tax



Note: Under the Design 1, the CO_2-tax is USD66 on the figure, which does not ensure exactly the same emission reduction in the electricity sector (between -18.39% and -18.86%).

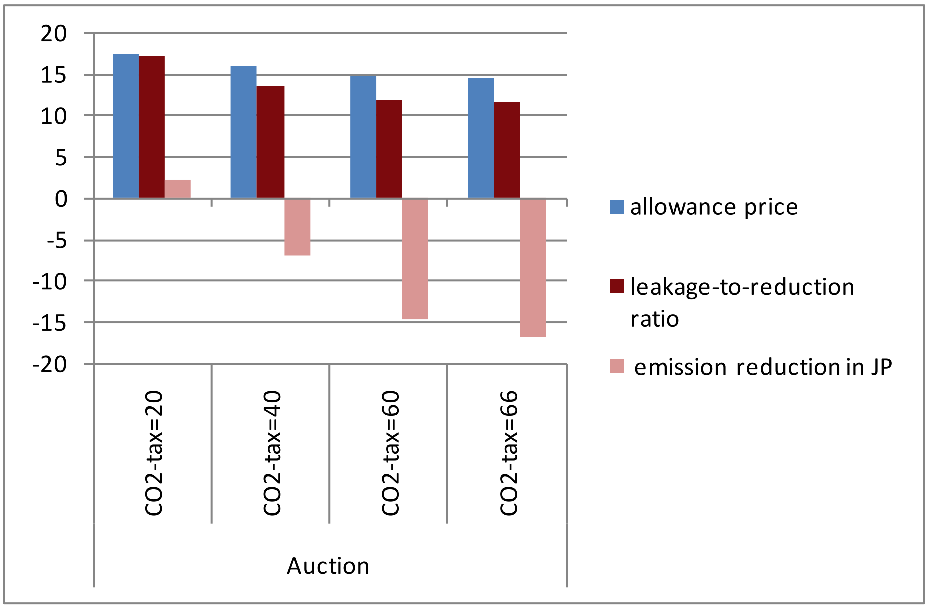
Another consequence of the higher allowance price under Design 2 is that the industrial sectors will further increase the product prices, while the electricity sector pays less per tCO_2e, hence increases less the electricity price (see Figure 12). There is then a trade-off between limiting the increase of the electricity price or of the industrial product prices.

The ordering of the allowance price among the scenarios is the same under both designs, but the levels vary. Indeed, the allocation mode plays a crucial role in determining the price level. The allowance price is lowest under "Auction" and highest under "OBA". The allowance price is higher under the "BA" scenario than under "Auction", because border adjustments limit the substitution of Japanese production by foreign production, which is one way of reducing CO_2 emissions. Hence a higher allowance price is needed to get lower unitary emissions. The price is higher under "OBA" than under "BA" because free allocation constitutes a subsidy to the production of polluting goods, which increases the demand for allowances, and hence the CO_2 price. This leads to higher production levels than under the "Auction" and "BA" scenarios (Figure 14). Consequently, to generate the same aggregate emission levels under "OBA" as in the other scenarios, lower CO_2 emissions per unit produced are required, which implies a higher CO_2 price.

It is interesting to examine the impact of increasing the CO_2 tax on the allowance price when the electricity sector is not included in the ETS (Design 2). As is apparent in Figure 8, when the CO_2-tax increases, the emission reduction of all sectors is larger, while the allowance price in the ETS as well as carbon leakage are lower.

[9] In Asuka et al. (2009) the exogenous allowance price is 3000Yen/tCO_2e, about USD 27/tCO_2e by applying the 2005 change rate.

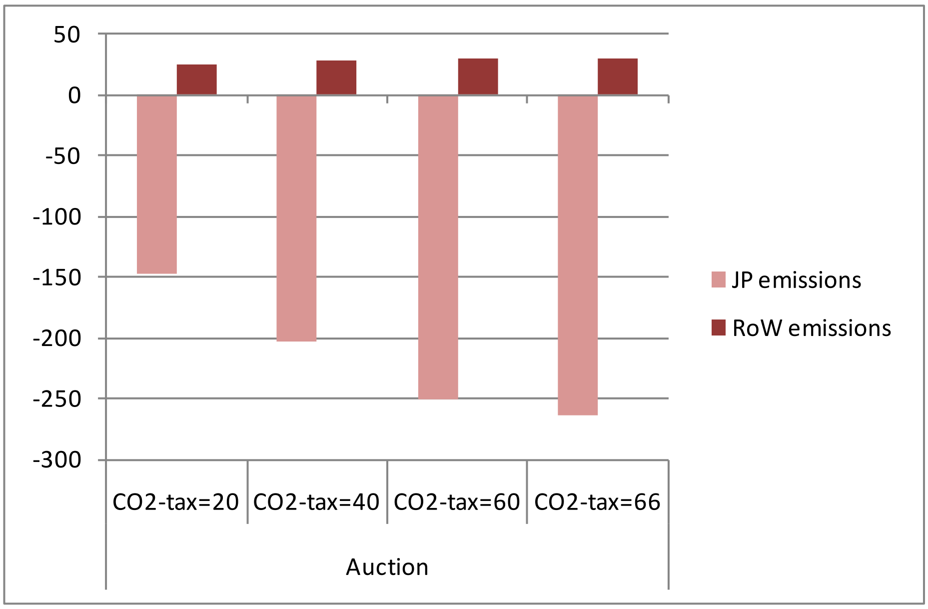
Figure 8: Impact of an increasing CO_2-tax on global emission reduction, ETS CO_2 price and leakage



These results are rather surprising because an increase in carbon leakage could be expected due to a higher increase of the electricity price. Indeed, the leakage-to-reduction ratio divides the emission variation in the rest of the world by the emission variation in Japan. When the CO_2-tax is increased in Japan, the emissions in the rest of the world increase, but the emissions reduction in Japan is more pronounced as illustrated in Figure 9. This example illustrates the limited information provided by this type of ratio.

The decline in the allowance price comes from the increase of the electricity price, which pushes upward the prices of electricity-intensive products. As a consequence, the quantities of Japanese products sold in Japan and abroad decrease, with the effect that the specific emissions need to decrease less to meet the cap of the ETS, leading to a lower allowance price.

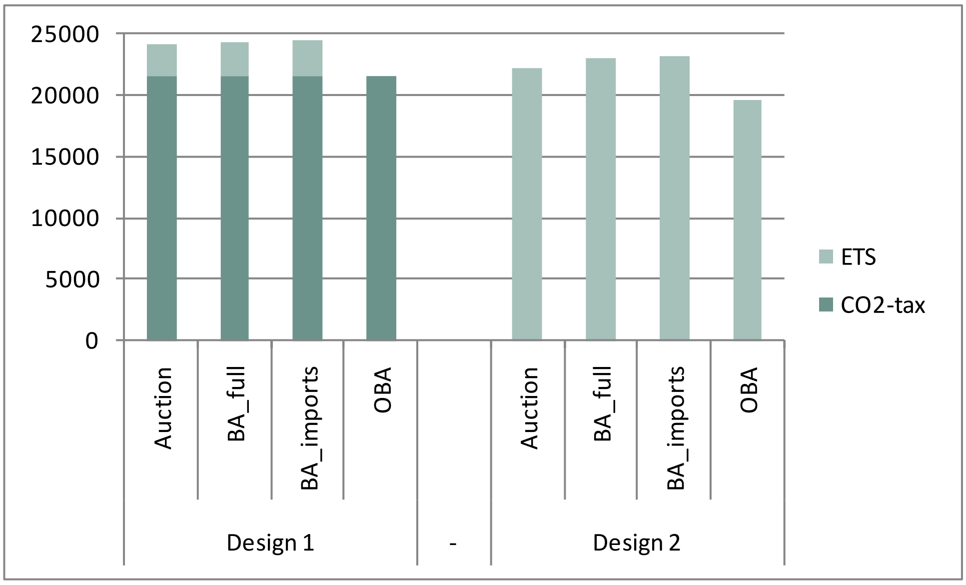
Figure 9: Emission variation in Japan and in the rest of the world between the "Auction" and "no-policy" scenarios



4.2 The potential of public revenues is large

Figure 10 presents the public revenues generated by the allowance auctioning and/or by the CO_2-tax. Under Design 1, more public revenues are generated since the biggest emitter, the electricity sector, pays more for each tCO_2e. Regarding all other features, the ordering among the scenarios is the same regardless of the design.

Figure 10: Public revenues from carbon pricing policy (million USD2005)

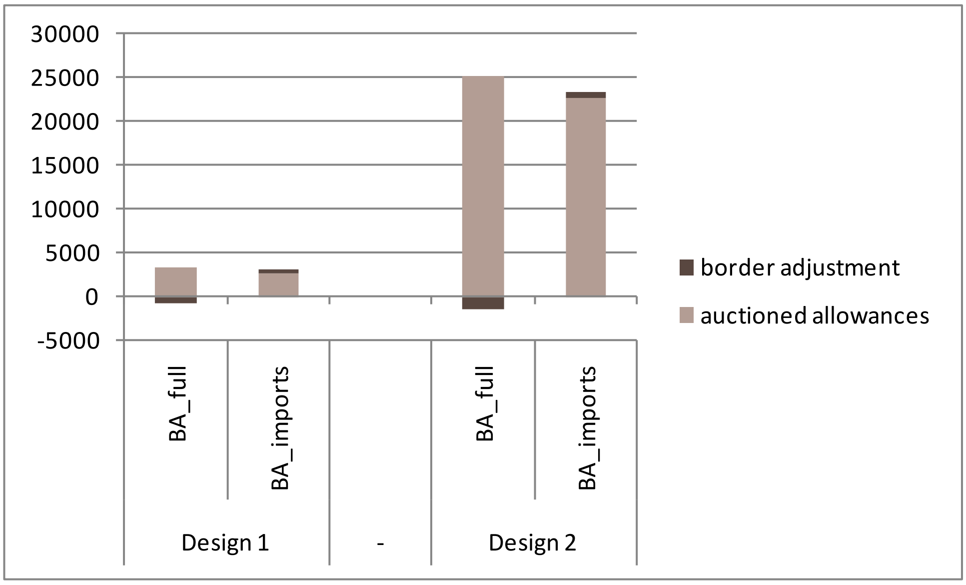


Note: Under Design 1, the CO_2-tax is USD66 in the figure. The revenues from ETS include the revenues or expenditures related to the BA if required.

The "BA" scenarios generate more revenues than the "Auction" and "OBA" scenarios. The proximity between the amount generated under "BA_full" and "BA_imports" hides in fact an important element. When the BA includes an export part, the mechanism induces public expenditures as is shown in Figure 11. Indeed Japan's exports of GHG-intensive products are a lot higher than its imports as is shown in Figures 3 and 4.

The "OBA" scenario generates the lowest amount of public revenues. Nevertheless, the collected money is still significant even though a large percentage of allowances is allocated for free. This is because the remaining allowances are sold at a higher price.

Figure 11: Public revenues from the ETS (million USD2005)

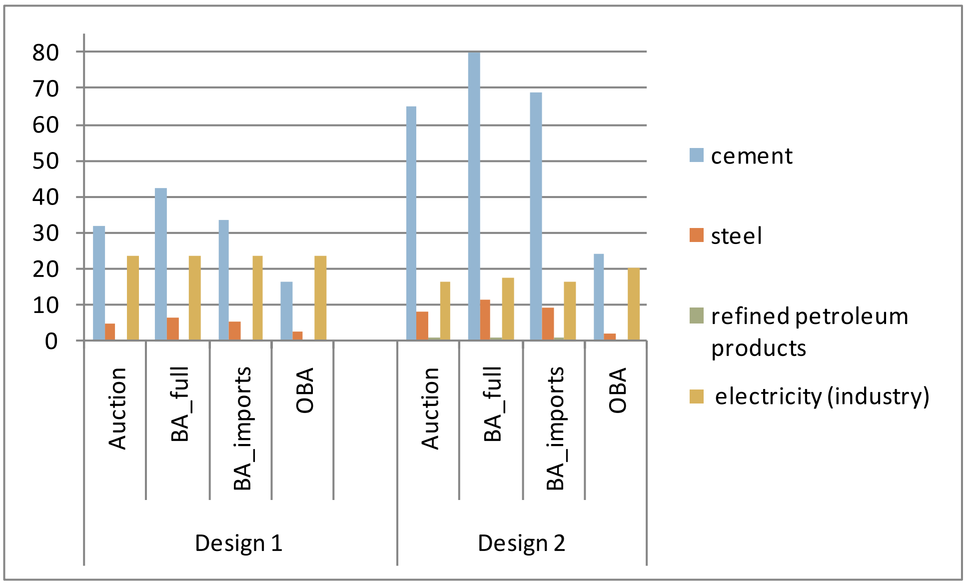


Note: Under the Design 1, the CO_2-tax is USD66 on the figure.

4.3 A trade-off between limiting the increase of the electricity price and the GHG-intensive product prices

As explained above, the choice of the design implies a different impact on the electricity price and on the industrial product prices. Indeed, under Design 1, the electricity price is higher than under Design 2. This affects the production costs of the industrial sectors but as these sectors have also access to less expensive allowances, the increase of product prices is limited as we can see on Figure 12.

Figure 12: Price change between the scenario with carbon pricing and the no-policy scenario (%)

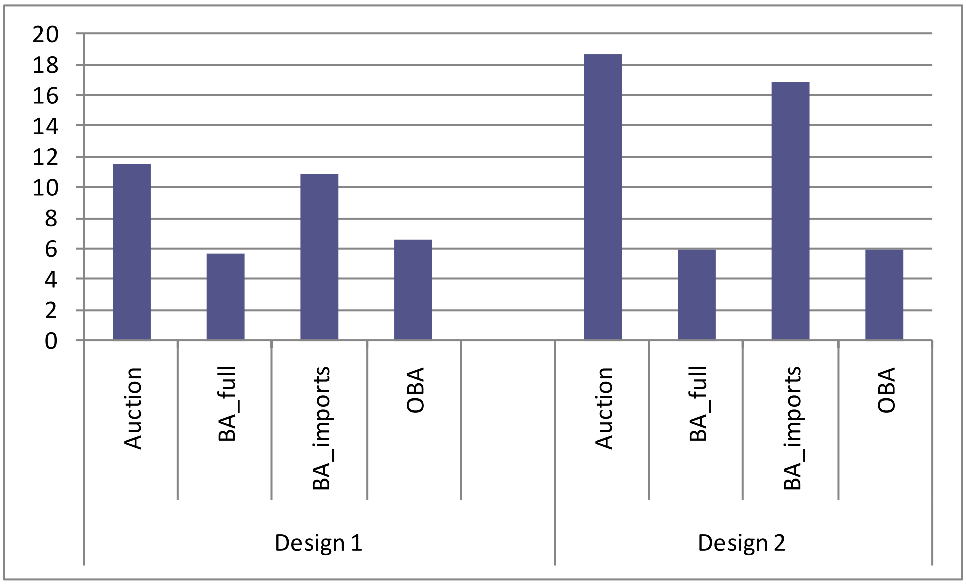


Note: Under the Design 1, the CO_2-tax is USD66 on the figure.

To exclude the electricity sector from the ETS allows preserving the exposed sectors from a high allowance price. Therefore, we could expect that requesting a bigger effort to be taken by the electricity sector allows preserving the exposed sectors and preventing large carbon leakage.

But, even under Design 2, carbon leakage can be limited in a significant way if the allocation mode is correctly defined. Figure 13 reveals that "BA_full" and "OBA" have similar performances to limit carbon leakage regardless the design. On the other hand, under "Auction" and "BA_imports", Design 1 outperforms Design 2 in terms of carbon leakage prevention.

Figure 13: Leakage-to-reduction ratio (%)

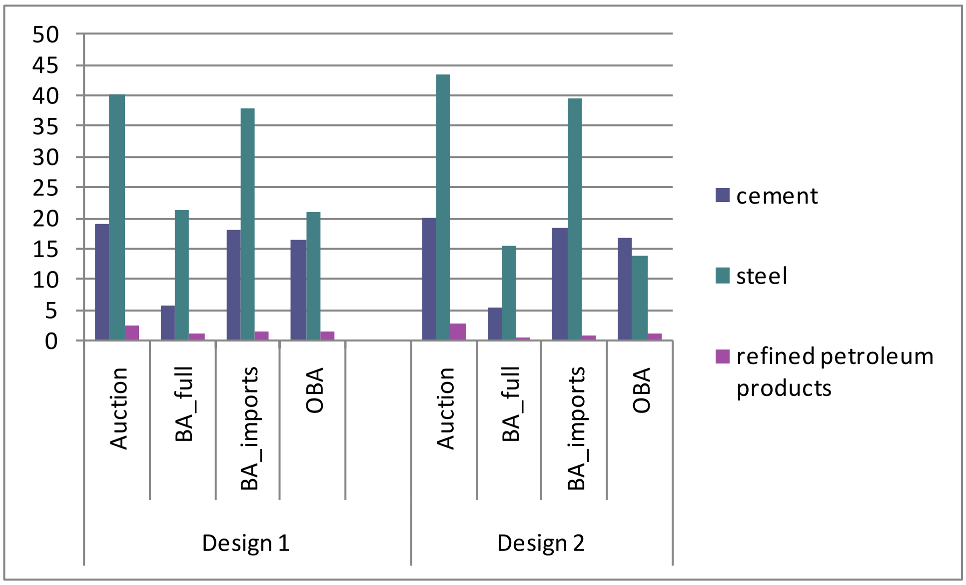


Note: Under Design 1, the CO_2-tax is USD66 in this figure.

Regardless of the design, a crucial result is the importance of BAs for exports in Japan. Indeed the leakage-to-reduction ratio under "Auction" and "BA_imports" are relatively close, revealing the poor performance to limit carbon leakage of a BA covering only imports. On the other hand, when exports are included in the "BA" scenario, carbon leakage is significantly reduced. This is because Japan is a big exporter of steel and even more so of cement, while its imports of these products are limited. Consequently, the challenge to limit carbon leakage is to limit the substitution of Japanese production by foreign production on foreign markets. This differs from results for the EU (Monjon and Quirion, 2011a).

Figure 14 disentangles the aggregated leakage-to-reduction ratio in order to examine the situation in each sector. The steel sector is the most exposed sector to carbon leakage with a ratio around 35 and 45% under "Auction" and "BA_imports". "BA_full" and "OBA" lead to similar decreases of the leakage-to-reduction ratio in the steel sector, while "BA_full" clearly dominates "OBA" in the cement sector. Lastly, the refined petroleum products sector seems little exposed to a carbon leakage risk.

Figure 14: Leakage-to-reduction ratio in the different sectors (%)

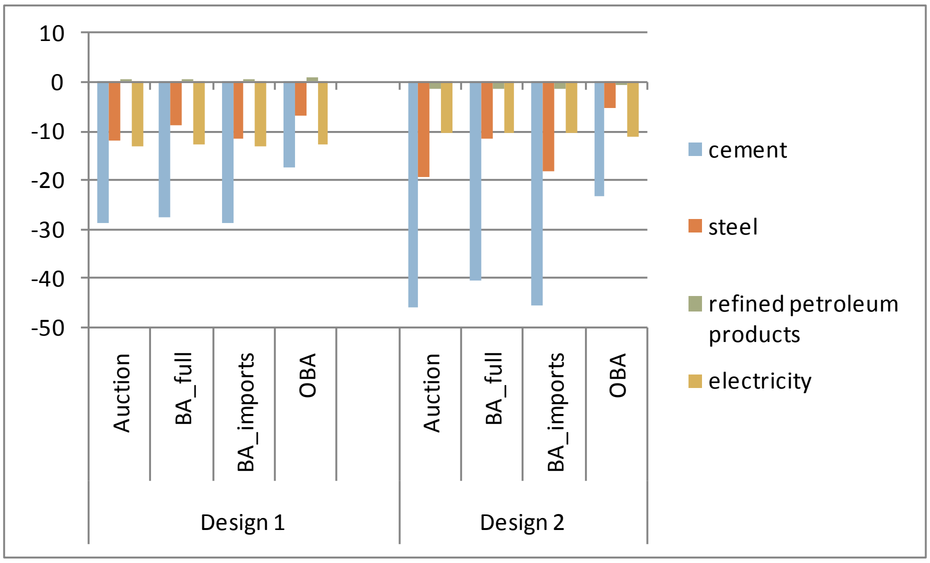


Note: Under the Design 1, the CO_2-tax is USD66 on the figure.

Finally, Figure 15 displays the variations in production levels in Japan relative to the no-policy scenario. The results differ depending on the allocation mode and on the design. While the output-based allocation mode performs relatively well in limiting the decline in Japanese production levels, "Auction" and "BA" induce large losses, in particular under the design 2.

The differences can be important among the sectors. The cement sector is the most impacted whatever the scenario, while the refined petroleum products sector is relatively little affected. The cement sector is very vulnerable to a high allowance price, leading to a better performance in limiting the production loss under of Design 1.

Figure 15: Japanese production level change between the scenario with carbon pricing and the no-policy scenario (%)



Note: Under the Design 1, the CO_2-tax is USD66 on the figure.

5 Summary and Conclusions

When implementing a mandatory ETS the Japanese government will have to define precisely the instrument. This analysis highlights the key role of some elements of an ETS.

1. **The inclusion or the exclusion of the electricity sector in the ETS is a politically crucial choice.** If the electricity sector is involved in the ETS, its allowance demand will push the allowance price, increasing the constraint on the industrial sectors. At the same time, the inclusion of the electricity sector in the ETS gives incentives ot ease the carbon constraint intensity a little for the industrial sectors.
2. The choice whether to integrate the electricity sector leads to **a trade-off between limiting the increase of the electricity price and the industrial product prices.**
3. **The emission reduction examined leads to a high CO_2 tax or to a high allowance price**, compared to those expected in the EU ETS for 2020.
4. **A BA efficiently limits carbon leakage if it covers imports and exports.** However, if the BA is only on imports, the leakage is about the same as without any BA.
5. **A BA covering imports and exports and an OBA both have a similar impact on limiting carbon leakage: both lead to a leakage-to-reduction ratio of around 6%.** Without any anti-leakage provision, excluding the electricity sector from the ETS limits a little the carbon leakage from the industrial sectors.
6. **The steel sector is the most exposed to a carbon leakage risk.** In this sector, carbon leakage is reduced the most when the electricity sector also covered by the ETS, with a slightly better performance of the OBA compared to a full BA. **However, in the cement sector, the carbon leakage remains almost as high as with full auctioning.**
7. **The potential public revenues from the ETS implementation are large.** The largest revenues are obtained if the electricity sector is targeted with a CO_2 tax. Free allocation based on output significantly decreases these revenues.
8. **The ETS implementation leads to a significant production decrease in the steel and cement sectors even with a full BA.** On this point, OBA is the most efficient option to limit the production decrease in the cement and steel sectors. Moreover, the production decrease is limited the most if the electricity is not included in the ETS.

The analysis highlights several important trade-offs for the final choice of the ETS design in Japan. We show that the different possible schemes have different impacts on the prices and on the production level in the different sectors. Moreover, the efficiency of the different anti-leakage options depends on the structure of the scheme. Lastly, there is no design which is superior to all other ones. Consequently, the political arbitration will play a particularly important role to decide on the design elements.

Annex 1: Description of the model

Consumption

In each region r={JP, RoW}, the representative consumer is assumed to have a two-tier utility function. The upper tier is a (logged) Cobb–Douglas function of the utility derived from consuming the goods produced by each industry, giving rise to fixed expenditures shares (αri) out of income (Yr):

$$U_r = \prod_{i=\{C,S,RPP,E\}} (u_r^i)^{\alpha_r^i} . Z_r^{(1-\sum_{i=\{C,S,RPP,E\}} \alpha_r^i)}$$

where α_r^i is the expenditure share of the region r in industry i, u_r^i is the sub-utility from the consumption of the varieties produced in the industry i and Zr represents the consumption level of the good. Indexes C, RPP, S and E represent cement, refined petroleum products, steel and electricity respectively.

Expenditures in region r in goods produced by industry i are then $\alpha_r^i . Y_r$. We assume that the expenditure parameters stay constant between 2005 (year used to calibrate the model) and 2020 (year used for the simulations of the business as usual and the different climate policies). GDP Yr is exogenous and growing.

In turning to the lower-tier of the utility function, we examine expenditure allocation in the industries C, RPP and S, each consisting of a domestic variety and a foreign variety. The sub-utility is a constant elasticity substitution (CES) aggregate of the two varieties. The representative consumer has different preferences over varieties depending on their places of production, allowing in particular for home bias. This preference parameter in region r for the domestic variety is denoted pref_{rr}^i while the preference parameter for the imported variety is denoted $\mathrm{pref}_{rr'}^i$, where r and r'={JP,RoW} and r'≠r. The sub-utility function is then:

$$u_r^i = ((\mathrm{pref}_{rr}^i . Q_{rr}^i)^{\frac{\sigma_i - 1}{\sigma_i}} + (\mathrm{pref}_{rr'}^i . Q_{rr'}^i)^{\frac{\sigma_i - 1}{\sigma_i}})^{\frac{\sigma_i}{\sigma_i - 1}}$$

Where i={C,RPP,S}, Q_{rr}^i (resp. $Q_{rr'}^i$) is the consumption level in region r of the good produced by industry i in region r (resp. r') and σi represents the elasticity of substitution (the Armington elasticity) between domestic and foreign varieties in industry i.

The electricity demand in region r is then the sum of the demand from the cement, refined petroleum products and steel firms localised in region r and of a fixed expenditure share out of income, $\alpha_r^E . Y_r$, from the representative consumer.

Supply

The CES specification of the representative consumer's utility has mostly been used in monopolistic competition models following Dixit and Stiglitz (1977) and Krugman (1980) where firms do not take into account the effect of their behaviour on other firms. Strategic interactions are therefore neglected, which is not very relevant for the industries analysed in this paper since they feature a small number of large firms. Consequently we explore the case where firms compete in quantities, as in a standard Cournot oligopoly. Thus, our modelling framework encompasses both the standard Cournot oligopoly (the substitution elasticity between the imported and the domestic variety tends toward infinity) and the pure competition Armington framework (if the number of firms tends towards infinity).

In the cement, refined petroleum products and steel sectors, each firm sells in both regions. In each region, there are nir domestic firms in competition. Firms are in competition regionally and, less intensively, internationally. Trade between the regions entails a constant per-unit transportation cost. Then the profit function of a firm localised in region r is:

$$\pi_r^i = (p_{rr}^i - mc_r^i) . q_{rr}^i + (p_{r'r}^i - mc_r^i - tc_{r'r}^i) . q_{r'r}^i - FC_r^i$$

where r and r'={JP,RoW} and r'≠r, i={C,RPP,S}, p_{rr}^i and $p_{r'r}^i$ are the delivered prices of the good produced by a firm of industry i localised in region r and sold, respectively, in region r and in region r', mc_r^i (resp. FC_r^i) the marginal (resp. fixed) production cost of firms localised in region r, q_{rr}^i (resp. $q_{r'r}^i$) the quantity sold in the domestic market (resp. in the foreign market) and $tc_{r'r}^i$, the (unit) transportation cost from region r to region r'.

This framework allows firms to set different prices in each market. This contrasts with the Dixit-Helpman-Krugman model in which firms perceive the same elasticity of demand in each market and therefore set export prices (net of transport costs) equal to their domestic prices (Head and Ries, 2001).

Each firm sets its production for domestic and foreign markets to maximise its profit, under quantity competition with the firms of the same region and of the other region. To determine the number of firms in each region, we assume that free-entry sets profits equal to nil in both regions. At the equilibrium, all firms from the same region are symmetric.

Excluding expenditures related to the climate policy, production costs (variable and fixed) are assumed constant but differ across regions.

Calibration and simulations

The model has been calibrated on 2005 data (prices and quantities). The values of the preference coefficient and the unit transport costs are determined by the calibration and are supposed to be constant.

Concerning the values of the Armington elasticity, large differences exist across sectors and countries. Moreover, estimates for Japan are rare. The larger the Armington elasticity, the more easily imported commodities may substitute for domestic commodities.

The strategy has been to use middle values of Armington elasticity: 2 for cement, 3 for refined petroleum products and 4.5 for steel.

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We are grateful for funding from the government of **Australia** and **Switzerland**, Agence de l'environnement et de la maîtrise de l'énergie (ADEME) in **France**, Deutsche Gesellschaft für Technische Zusammenarbeit (GTZ) and Ministry of Environment in **Germany**, Ministry of Environment in **Finland**, Ministry of Foreign Affairs (MFA) in **Norway**, Swedish Energy Agency (SEA) in **Sweden**, Department for Environment, Food and Rural Affairs (DEFRA), the Office of Climate Change (OCC), Department of Energy and Climate Change (DECC), Department for International Development (DFID) in the **UK**, The Carbon Trust, Nordic COP15 Group, Corus Steel, Holcim, Ministry of Environment (MOE) in **Japan**, European Climate Foundation (ECF) in The **Netherlands** and the German Marshall Fund of the **United States**.