Department of Economics- FEA/USP

Winners and losers: the distributional impacts of a carbon tax in Brazil

MARIA ALICE MOZ-CHRISTOFOLETTI

PAULA CARVALHO PEREDA

WORKING PAPER SERIES Nº 2021-08

DEPARTMENT OF ECONOMICS, FEA-USP

WORKING PAPER Nº 2021-08

Winners and losers: the distributional impact of a carbon tax in Brazil

Maria Alice Moz-Christofoletti (maria.christofoletti@usp.br)

Paula Carvalho Pereda (pereda@usp.br)

Abstract:

Through its NDC, Brazil pledged to reduce its GHG emissions by 43% below 2005 levels in 2030, respectively. Carbon pricing could play a key role in meeting this objective. However, a range of issues can emerge when introducing it. Among these issues, the distributional impact has been frequently highlighted as an obstacle to the public acceptance of such a mitigation policy. This paper examines the short-run welfare and emission effects of an economy-wide carbon tax on Brazilian households. The distributional impact is examined by estimating the tax burden relative to annual expenditures and changes in total GHG emissions across income levels, using tax rates consistent with the Paris Agreement and considering a lump-sum rebate that keeps the government revenue neutral. For this, we calculate energy-related GHG emissions coefficients from fossil fuel burning for the whole household consumption basket, and price and expenditure elasticities which account for the zero-expenditure and underdeclaration problems. Our results indicate that the incidence of the carbon tax is effective in reducing emissions in the short run, but imposes welfare losses, especially on the poor. The consideration of compensation mechanisms is critical when designing this type of environmental tax, specially in the context of a highly complex tax-system.

Keywords: Carbon Taxation; Censored QUAIDS; Hybrid Input Output

JEL Codes: H23;K32; O13

Winners and losers: the distributional impacts of a carbon tax in Brazil

1 Introduction

Since 2005, reductions in land-use and forestry emissions have contributed to the expressive decrease of the overall GHG emissions in Brazil. However, the steadily decline of the country's share of renewable sources in the energy mix boosted energy-related emissions: currently, the sector emits 30% more compared to 2005 levels (BRASIL, 2019). Energy emissions are projected to rise given the recent oil discoveries in the offshore fields and the near exhaustion of the country's environmentally feasible hydropower potential.

Environmental economists have been advocating for carbon taxes as the fastest and most efficient instrument to curb emissions from fossil fuels[1]. The idea is to transfer the environmental costs paid by third parties (society) to those that are responsible for them (polluters)[2]. Carbon tax might also have the potential to generate dividends (Nordhaus, 1993; Pearce, 1991; Goulder, 1995; Fullerton and Metcalf, 1997): the reduction of the environmental damage ("first dividend") and the potential to use its revenue to reduce other distortionary taxes ("second dividend"), such as taxes on labor and capital[3], or to support funding investments on cleaner power generation, smart vehicles, and improvements in energy efficiency, keeping the government budgetary position and the overall tax burden unchanged (also known as "revenue neutrality")[4].

Despite the increase in jurisdictions considering carbon pricing initiatives (World Bank, 2020), the number of underdeveloped and developing countries that implemented carbon taxation is still low compared to developed countries[5,6]. Even with Brazil's

---

[1] Compared to other carbon pricing mechanisms, carbon taxes are also considered a relatively simple instrument to impose on emitters, particularly in settings with a large number of small emission sources - such as transportation - as it lowers transaction costs (Carattini, Carvalho, and Fankhauser, 2018).

[2] In 2019, 57 carbon pricing schemes, such as carbon taxes or cap-and-trade systems, had been established or planned, covering 20 % of the global GHG emissions (Ramstein et al., 2019).

[3] Goulder (1995) defined the two types of double dividends: i) weak - which states that recycling environmental tax revenues through lowering distortionary taxes leads to cost savings compared to the case where revenues are returned via lump-sum transfers and ii) strong - where a revenue- neutral substitution of a green tax for typical or representative distortionary taxes produces zero or negative welfare gross cost.

[4] Globally, an estimated 4% of carbon tax revenues have been used to lower other taxes, 28% for general funds, and 15% for environmental mitigation spending (IMF, 2019).

[5] Carbon taxation was first implemented in Finland, Poland, Sweden, Norway, Denmark, Latvia and Slovenia in the 1990s. Other countries/provinces have been recently adopted this instrument, such as Estonia (2000), Switzerland (2008), British Columbia (2008), Ireland (2010), Iceland (2010), Japan (2012), France (2014), Mexico (2014), Portugal (2015), Argentina and Chile (2018), Canada (Federal Carbon Price) (2019) and South Africa (2019), with a wide range of tax rates, coverages, exemptions and revenue recycling schemes. For a complete description of current price schemes, see Ramstein et al. (2019).

[6] According to World Bank (2020), national carbon tax have been implemented or scheduled for implementation in Argentina, Mexico, Ukraine, Kazakhstan, Colombia and South Africa vis-a- vis Chile, Canada, EU-27, United Kingdom, Switzerland, Liechtenstein, Iceland, Norway, Japan, Australia and New Zealand.

National Policy on Climate Change (PNMC)[7] proving the possibility of using fiscal and tax measures to reduce anthropogenic GHG emissions, the country has used very little taxation (and sometimes subsidies) to penalize activities with negative environmental externalities: in 2017, 94% of the country's emissions came from non-taxed polluting activities, the largest percentage in the world (OECD, 2018)[8,9]. Benefits to oil and gas producers include special tax incentives for infrastructure development in various regions, as well as a special tax regime for equipment used in the exploration and development of hydrocarbons, and exemptions for coal used in electricity generation. This incipient uptake and acceptability of carbon taxes as a potential policy in developing settings can be linked to concerns about equity effects: critics of the double dividend existence (Fullerton and Metcalf, 1997; Babiker, Metcalf, and Reilly, 2003) state that, under certain circumstances, a shift to environ- mental taxes may increase the burden of the tax system. Therefore, distributional burdens of a carbon tax among different classes of households or production factors in a particular country needs to be assessed, not assumed (Bowen, 2015; Fullerton, 2011; Fullerton and Muehlegger, 2019).

Since a comprehensive tax system reform is a top-priority of the current Brazilian government and that a concrete proposal is still not defined, it is critical to shed light on potential distributional effects of an economy-wide carbon tax given the country's historical tax burden and complex taxation structure. In this paper, we address this point by focusing on welfare and emissions outcomes of a hypothetical carbon tax on goods and services consumed by Brazilian households. To analyze the carbon tax effects, we use a top–down approach linking macro and micro models: first, direct and indirect GHG emissions coefficients from fossil energy- related fuels burned are calculated to all household basket' goods using a hybrid input-output model (I-O); second, household consumption patterns across different income-levels are identified using a system of demand equations, circumventing the zero-expenditure and under-declaration problems through a censored model with instruments for expenditures and prices; and third, the distributional impact is examined by looking at the tax burden relative to annual expenditures and changes in GHG emissions for richest and poorest households, using

---

[7] The PNMC is the regulatory framework that guides the government under the climate change institutional arrangement since 2009, being promulgated through Law 12,187 of 29 December 2009.

[8] The Brazilian government offers a range of tax and budgetary subsidies for fossil fuel production, which amounted to R$ 11.6 billion (USD 4.9 billion) in 2015 (Nuaimy-Barker, 2015).

[9] At the federal level, the only Brazilian tax that seems to have an environmental purpose is CIDE-Fuels, implemented in 2011. However, CIDE-Fuels is far from being a carbon tax because i) it applies only on gasoline and diesel, and ii) its revenue has been managed to stabilize fuel prices or to subsidize fossil fuels and finance transportation infrastructure programs, rather than discouraging fossil fuel consumption. According to the government estimates, the current rate of R$ 100.00/m^3 and R$ 50.00/m^3 levied on gasoline and diesel are equivalent to an implicit carbon tax of USD 13.70/tCO_2e and USD 5.90/tCO_2e, respectively, rates considerably below the range of USD 40/tCO_2e to USD 80/tCO_2e needed to be consistent with the Paris Agreement. Nonetheless, better examples are found at the state and municipal levels, such as the Ecological Sales Tax (*ICMS Ecológico* or ICMS-E). It aims to reduce the economic gains from deforestation and encourage environmental preservation by distributing the revenues obtained from the sales of goods and services according to the reduction in deforestation and percentage of area occupied by protected areas, indigenous lands and *quilombolas* - traditional communities composed of the descendants of runaway slaves.

tax rates consistent with the Paris Agreement (USD 40/tCO$_2$e and USD 80/tCO$_2$e).

Taking into account the complexities of the Brazilian tax-system and the double dividend hypothesis, we analyze a revenue-neutral carbon tax - to prevent an increase of the tax burden - considering different rebate options currently considered as part of the Brazilian tax reform (lump-sum)[10]. In particular, we explore the implications of recycling carbon tax revenues from richest directly to poorest households in the form of social tariffs for the consumption of energy goods. The expansion of direct transfers is one of the current proposals of the Brazilian tax reform (mainly, the substitution of the "*Bolsa Familia*" by the "*Renda cidadã*"), but their criticism lies on the identification of the respective revenue sources. Sensitivity analysis is also conducted with narrower tax bases depending on the carbon content of goods/products, as well as using social tariffs.

Our results indicate that the first-dividend could be observed if an economy- wide carbon tax is implemented, as it is effective in reducing emissions by up to 4.2%. However, this instrument imposes higher welfare losses on low-income households (0.06% and 0.10% in relation to total expenditures for richest and poorest households, respectively). Narrowing the tax base only on products and services with high carbon emissions might reduce the regressiveness of the carbon tax, as well as using social tariffs using richest households' revenues. Therefore, our findings suggest that compensation mechanisms are critical and need to be considered when designing a carbon tax, especially in the context of a highly complex tax-system.

Given that the empirical literature on the distributional impacts of carbon taxation indicates ambiguous results11, our findings reinforce the empirical evidence of small regressive effect based on lump sum redistribution in the context of developing countries (Wang et al., 2016; Ohlendorf et al., 2018). Particularly for Brazil, other studies also suggest that this type of taxation is regressive in spite of being efficient in terms of emission reduction (Magalhães and Domingues, 2013; Silva Freitas et al., 2016; Grottera, Pereira Jr, and La Rovere, 2017).

We also add to the empirical literature a methodological contribution by combining a more flexible approach with household microdata, allowing for both an in-depth

---

[10] The literature also suggests that, the more salient the benefits from recycling options are, the better the acceptability of a carbon tax is. Therefore, depending on the context, recycling carbon pricing revenues as lump-sum dividends can generally be a good strategy: lump-sum dividends are highly salient, create constituents in favor of climate policy and could also be favorable in political climates marked by solution aversion or lack of political trust (Klenert et al., 2018).

[11] Many studies have found an overall tendency for regressive impacts, especially in developed countries such as Denmark (Wier et al., 2005), Spain (Tomás, López, and Monsalve, 2020), Sweden (Brannlund and Nordstrom, 2004), Netherlands (Kerkhof et al., 2008), France (Bureau, 2011), United States (Grainger and Kolstad, 2010; Fullerton, 2011) and United Kingdom (Feng et al., 2010). However, developing countries have shown an inconsistent picture, with a tendency towards proportional or progressive impacts (Wang et al., 2016; Ohlendorf et al., 2018). In this sense, these studies also show that recycling the carbon tax revenue either through a lump sum or direct transfers/subsidies could help alleviate the potential regressivity of a carbon tax or even convert it to a progressive carbon tax. Nevertheless, progressive impacts have also been found in developed settings such as Italy (Tiezzi, 2005) and British Columbia (Beck et al., 2015).

understanding of families' substitution patterns between carbon intensive and non-intensive goods and services, as well as assessing the distributional effects of different carbon tax policy designs on welfare and emissions. This is particularly important for the Brazilian current political debate around the tax reform and consequences of carbon pricing instruments, as the recycled policy options assessed in this paper are also being considered for future implementation. Beyond these tax reform discussions, Brazil's Nationally Determined Contribution (NDC) is also expected to be revised this year[12] and, unlike other countries[13], it did not present in the document any prospect of participation in an international carbon market or carbon tax as mechanisms to achieve its mitigation targets [14], despite having its regulatory framework allowing for it. Therefore, the government might consider the implementation of market instruments to meet the country's mitigation targets and reduce overall mitigation costs.

The paper is organized as follows: first, we present the literature review of the distributional effects of a carbon tax in Section 2; the empirical strategy and description of the data sources used in this study are presented in Section 3; in section 4 we discuss our empirical results; and in Section 5 we summarize our conclusions and policy implications.

2 Literature Review

There are winners and losers when a carbon tax is introduced in an economy (Fullerton, 2011; Fullerton and Muehlegger, 2019; Bowen, 2015; Wang et al., 2016; Cronin, Fullerton, and Sexton, 2019). The overall distributional effect of this instrument is complex and influenced by many factors - such as the household consumption patterns, production structures and firm competition, distribution of co-benefits from improved environment quality[15]and the carbon tax design. The latter, in particular, includes a number of important factors, each of which has implications in terms of effectiveness and distribution: who should pay the tax, what should be taxed, how much is the tax rate and the use of tax revenue (also called preferential policy design).

Wang et al. (2016) and Ohlendorf et al. (2018) provided a very comprehensive literature review on distributional impacts of carbon taxes, particularly on different

---

[12] Brazil pledged to reduce its overall GHG emissions by 37% and 43% below 2005 levels in 2025 and 2030, respectively.

[13] 190 parties submitted climate strategies for the 2015 Paris Agreement. Most strategies include objectives for both mitigation (reducing emissions) and adaptation (building resilience to climate change) (IMF, 2019).

[14] As stated in the NDC document, "Brazil reserves its position regarding the possibility of using any market mechanisms that may be established under the Paris Agreement" (BRASIL, 2015).

[15] Carbon pricing can produce significant environmental co-benefits, for example, reduced air pollution from coal combustion and externalities like reduced congestion from motor vehicles, at least until these other externalities are fully priced through other policies. Co-benefit estimates can be quite large, averaging USD 57.5 per tonne of CO_2 across the top-twenty emitters, though with substantial cross-country variation (e.g., due to sharp differences in population exposure to pollution) (Parry, Veung, and Heine, 2015).

income groups of households. In general, existing studies focus mainly on the cost distribution and observe regressive impacts of an economy-wide carbon tax in higher-income countries, even when considering both direct and indirect impacts - that is, accounting for emissions related to fossil fuel use and production of goods and services for final consumption, as well as substitution and income effects at the household level. This overview is consistent with the Intergovernmental Panel on Climate Change (IPCC) (Edenhofer, 2015), which indicates that the impacts of national carbon taxes on consumers would likely be somewhat regressive in high- income countries.

For example, using an I-O model and applying a direct and indirect average tax of e81/year and e35/year in Denmark, Wier et al. (2005) found that low-income families paid carbon taxes constituting around 0.8% of disposable income, while high-income families paid approximately 0.3% of disposable income. In Sweden, Brannlund and Nordstrom (2004), based on a household demand model (Quadratic Almost Ideal Demand System (QUAIDS)) , observed that poorest and richest households experience a welfare loss of 0.52% and 0.33% of their disposable income, respectively, for a 100% increase of the CO_2 price, which was based on USD 46/tCO_2. The regressive nature of CO_2 taxes in both studies can be explained by two fac- tors: i) CO_2 intensities vary strongly between consumption goods, with food and transport being very CO_2 intensive, and services and financial transfers being at the other end of the scale; and ii) low-income cohorts mainly consume carbon-intensive necessities, while high-income cohorts spend a large part of their income on "luxury" items that have a higher service component. Since carbon taxes in Denmark and Sweden were introduced in 1992 and 1991, respectively, the findings from these studies correspond to *de facto* effects. Simulations for Spain from Tomás, López, and Monsalve (2020), based on an environmentally extended I-O model, also indicated an effect disproportionally higher on people of small municipalities and rural areas.

Using an I-O model combined with consumer expenditure survey for the U.S., Grainger and Kolstad (2010) observed that, for a tax of USD 15/tCO_2, the poorest quintile's burden (as a share of annual income) is 3.2 times that of the wealthiest quintile. Similar results were also found by Hassett, Mathur, and Metcalf (2007), where the additional cost of a carbon tax was approximately 3.7% for the lowest decile, which is over four times the added burden of the highest decile. These studies suggest that the regressivity of the policy is driven largely by direct energy consumption.

However, progressive effects were found in empirical studies for Italy and British Columbia. Tiezzi (2005), through estimation of an Almost Ideal Demand System (AIDS) model and using household data from 1985 to 1996, verified that the welfare losses as a percentage of expenditure were 0.4% and 0.8%, respectively for Italian households in the lowest and highest expenditure levels. This might be due to the fact that the taxation mainly fell on transport fuels, whereas heating' fuel prices increased relatively less. The tax burden also seems to affect mainly households with one and two adults and decreases for larger families, which could be explained by the fact that the tax burden due to car ownership, for instance, is more distributed as the number of household members increases, because the number of cars owned does not increase linearly with the number of household members. Beck et al. (2015), using a static CGE model of the Canadian economy, estimated that a carbon tax of USD 30/t applied on all combustion GHGs would cause welfare losses of 0.2% and 0.6% for poorest and richest households, respectively. The progressive character of the tax would be enhanced by the introduction of revenue recycling measures - the poorest households would present an increase of 0.8% in welfare while richest households would have their welfare levels reduced by 0.2%.

In the limited number of studies focusing on developing economies, the results are more diverse. Combining emission information estimated via an I-O model, together with micro data from the Mexican National Survey of Household Incomes and Expenditure, Renner (2018) found that, in the case of the highest simulated tax rate of USD 50/tCO_2e and including CH_4 and N_2O in the taxation, the relative welfare losses would be 4.2% and 3.4% of total expenditures for the poorest and richest households respectively, while in the case of a carbon tax rate of USD 20/tCO_2 exclusively taxing CO2 from energy use, welfare losses would be progressive and account for around 1% of total expenditures for all households.

In contrast, based on an econometric model, Brenner, Riddle, and Boyce (2007) observed that with a charge set at 300 yuan/tCO_2[16], and with equal redistribution of the revenues, the effect of the carbon charge would be progressive in China: the lowest decile would pay 2.1% of total expenditures to satisfy the levy, and the highest decile would pay 3.2%. Using a 2012 Multi-Regional Input-Output for 30 Chinese provinces, Wang et al. (2019) estimated a regressive effect, especially on rural households, if domestic fuels are taxed; however, when taxing transport fuels, the authors found progressive impacts. Using a general-equilibrium (CGE) model for South Africa, Devarajan et al. (2011) indicated that, compared to other instruments, a direct tax on carbon emissions imposes the lowest distortion, where household welfare declines by roughly 0.3% in order to reduce emissions by 15%. On the other hand, an indirect tax on pollution-intensive commodities imposes a higher cost - by as much as 10 times that of a carbon tax. When assessing the results of a USD 30/tCO_2 tax for different income groups in 87 low- and middle- income countries (countries with per capita incomes below USD 15,000 per year (at PPP-adjusted 2011 USD)) and using a multi-regional I-O table, Dorband et al. (2019) found that carbon pricing has, on average, progressive distributional effects. Due to an inverse U-shape relationship between energy expenditure and income, the authors noted

---

[16] The rate was comparable to existing carbon charges in other countries in 1999.

that USD 15,000 (PPP-adjusted) is the turning point at which carbon pricing is likely to be progressive (and regressive above this threshold).

Results of previous studies for Brazil are also diverse. Grottera, Pereira Jr, and La Rovere (2017), considering a R$ 50/t tax on carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) emissions and using an input-output model for 2005, found that despite the markedly recessive effect when the revenue is not reinserted into the economy, the measure is progressive since income inequality decreases, especially if direct transfer to households is considered. However, when the revenue is used to reduce taxes on the labor factor, the carbon tax contributes to increase income inequality. Based on the same method and tax rate, Silva Freitas et al. (2016) estimated welfare losses measured by compensatory variation of 3.1% and 1.2% for lower and higher income deciles, respectively. Similarly, considering the same tax applied on GHG emissions and using a computable general equilibrium (CGE) model, Magalhães and Domingues (2013) found that the poorest and richest households would reduce their total consumption by 2.2% and 1.8%, respectively, in a scenario without any revenue recycling, but when considering recycling to households of that part of the revenues from the carbon tax, the reductions were 1.2% and 0.9%, respectively.

As observed, the tax rate simulated and implemented varies considerably among the countries. Wang et al. (2016) indicated that these rates can be either based on what is perceived as politically feasible, or based on the marginal abatement cost of carbon, or simply on budget requirements. In the last report prepared by the High-Level Commission on Carbon Prices (Stiglitz et al., 2017), it was concluded that carbon prices of USD 40–80/tCO_2 and USD 50–100/tCO_2 would be necessary in all countries to achieve the targets of the Paris Agreement in by 2020 and 2030, respectively. However, about three-quarters of emissions covered by existing carbon pricing schemes are priced at less than USD10/tCO_2e (Timilsina, 2018), suggesting that these tax rates are relatively low compared to what actually is needed to achieve internationally agreed climate targets in the long-run (Boyce, 2016; Pindyck, 2013).

In general, the literature shows that the selection of income or expenditure to measure the relative cost may also affect the results. The studies which focus on the general effects of carbon taxes on well-being are more likely to adopt measures of expenditure rather than income due to three reasons, according to Flues and Thomas (2015): i) current consumption measures the current standard of living better than current income, based on the premise that households derive utility from the consumption of goods and not from income (Ravallion, 1992); ii) expenditure is likely to be a better (though still imperfect) proxy for lifetime well-being than income and iii) adopting an expenditure base provides a more reliable picture of the lifetime distributional effects of a consumption tax because it removes the influence of borrowing and saving from the analysis. Therefore, measures of regressivity are often diminished when evaluated according to lifetime income or permanent income, or a proxy such as annual expenditures (Cronin, Fullerton, and Sexton, 2019).

Overall, the empirical assessments indicate that the revenue generated through the tax could also be used to counteract potential negative distributional impacts and potentially generate double dividends (Goulder, 1995; Fullerton and Metcalf, 1997). Some of the options for using the revenue earned from carbon taxes include ex-ante measures, such as public transport subsidies (Brannlund and Nordstrom, 2004), as well as ex-post measures, such as lump-sum transfers to households (Sajee wani, Siriwardana, and Mcneill, 2015; Brenner, Riddle, and Boyce, 2007), and relief of existing and naturally distorting taxes on labor, income or revenues (Callan et al., 2009; Pereda et al., 2019). The studies suggest that low income households would benefit more when carbon tax revenue is recycled as a lump-sum rebate than used to cut existing taxes. If the carbon tax revenue is used to cut existing taxes, higher income households will benefit the most (i.e., their welfare loss due to the carbon tax decreases) (Timilsina, 2018).

In theory, for efficiency and fairness purposes, the tax should be applied as broadly as feasible to all greenhouse gas emissions, regardless the source. In light of the practical experience of countries that have introduced a carbon tax, along with studies proposing and modeling a hypothetical carbon tax, Wang et al. (2016) stated that the tax generally is levied on fossil fuels from both primary (e.g., oil, coal, biomass) and secondary (e.g., electricity, fuel oil) energy sources. Broader coverage tends to be more cost-effective to reduce emissions, since the same marginal incentive for reductions is observed across a broad range of sources. They also have higher revenue potential and are considered simpler and fairer. Narrow coverage, on the other hand, is easier to measure and to enforce (particularly for downstream taxation, which is harder to monitor and enforce) and could exempt vulnerable/politically powerful sectors.

In addition, the positioning of regulation is also an important element in the design of a carbon tax. There often exist “upstream” or “downstream” choices in the energy chain to impose the tax, to minimize collection and monitoring costs and to ensure maximum coverage. In an upstream approach, refineries and importers would pay a tax based on the carbon content of their gasoline, diesel fuel, or heating oil; coal mine operators would pay a tax reflecting the carbon content of extracted coal; and natural gas companies would pay a tax reflecting the carbon content of their produced and imported gas. A downstream point of regulation or taxation would assign compliance responsibilities to the final emitters. Alternatively, a hybrid upstream-downstream approach could address a broad base, such as in a system that covers power plants’ direct emissions and transportation’s embedded emissions, with refineries serving as the point of compliance for petroleum fuels (Aidy, 2017). In the absence of the hybrid approach, Metcalf and Weisbach (2009) pointed out that imposing the tax upstream could cover a broader base (i.e., a larger fraction of an economy's emissions) and more economies of scale could be obtained in tax administration as there are fewer upstream producers than downstream consumers and the cost will be lower per unit of tax. Additionally, an advantage of an upstream system is that it treats all fossil carbon equally, regardless of where it is burned. Arguments for downstream (e.g., households or energy-using industries) imposition of the tax tend to be based on a claim that a downstream tax is more visible and, therefore will have a greater effect.

Much of the research on the distributional impact of a carbon tax relies on I-O models in combination with household expenditure surveys. The I-O model is used to analyze the direct and indirect consumer price changes caused by higher fossil fuel prices. Subsequently, these prices are combined with data from consumer expenditure surveys to estimate incidence of the carbon tax. However, this modeling approach misses some important elements that can have significant impacts on the incidence of the policy: it does not allow industries or households to change their behavior in response to increases in the price of carbon intensive commodities, and assumes an inherent homogeneity in the sector-by-sector technology. Additionally, possible constraints to the supply of production factors - such as labor and capital are not taken into account, and generally the models assume full pass-through of price increases from producers to consumers in the form of higher prices. Since it is a static analysis, these models present stocks at a given period of time. As a result, these models highlight the short run distributional outcomes rather than the dynamic effects of a carbon tax on production techniques and consumption bundles (Mathur and Morris, 2014).

A wide range of empirical studies on the distributional aspects of environmental policies also have used computable general equilibrium (CGE) models. In particular, studies which take into account recycling schemes for carbon pricing tend to use this approach. These models provide a higher degree of flexibility in choosing functional forms to represent agents' behavior and also allow substitutions between factors and inputs, as well as passing the burden of a tax forward to consumer prices or backward to factors. Despite that, studies using CGE models have found that consumption taxes are entirely passed forward to consumers (Metcalf and Weisbach, 2009), as expected under perfect competition. Boyce (2016) relaxed this assumption by allowing some of the cost to fall on producers and, ultimately, stock owners, which makes a carbon tax less regressive. In addition, econometric models have also been used to assess the distributional impacts of a carbon tax, through the estimation of a consumer demand system, generally in combination with the I-O approach. By assessing behavioral response, these models tend to be more flexible, allowing complementarities and substitution relationships among the goods, which can improve the identification of the distributional effects. Commonly used econometric models are almost ideal demand systems (AIDS) (Tiezzi, 2005), their more flexible quadratic specification (QAIDS) (Brannlund and Nordstrom, 2004; Nikodinoska and Schroder, 2016) or more recently the exact affine Stone index (EASI) demand system (Reaños and Wölfing, 2018).

3 Empirical Strategy and Data

An analysis of the distributional emission and welfare consequences of a carbon tax requires detailed data on households' carbon footprints and a clear understanding of their consumption behavior. First, we calculate GHG emission coefficients for several products and services consumed by Brazilian households using a hybrid input-output (HIO) approach. Then, for a nationally representative sample of Brazilian households, we calculate expenditure and price elasticities for these goods and services using a censored demand system. Finally, we combine these estimates to measure the welfare and emission effects of a carbon tax.

3.1 Emission coefficients: input-output model

We use the national input–output matrix for 2010, built according to Guilhoto (2010). This matrix is constructed based on the 2010 Supply-Use Tables (SUTs) provided by the Brazilian Institute of Geography and Statistics (IBGE) and contains information on production and intermediate consumption, in monetary units, of 128 products and 68 economic sectors. To obtain the carbon footprint of goods and services, we develop a hybrid matrix ($\boldsymbol{E}_{exn}$) containing the amount of energy consumed - expressed in physical units (tonnes of oil equivalent, toe) of e sources of energy in n economic sectors ($e < n$) - by each sector, based on information from the Brazilian Energy Balance (BEN). The matrix $\boldsymbol{E}_{exn}$ provides energy requirements (in toe) for 21 economic sectors from 24 energy sources [17].

We follow Montoya, Lopes, and Guilhoto (2014) and Grainger and Kolstad (2010) to reconcile the energy sources from BEN with the products from SUTs. Then, in the matrix $\boldsymbol{Z}$, which represents the inter-industrial transactions -, we substitute the monetary flow by the energy intermediate input flows, creating the hybrid matrix $\boldsymbol{Z}^*$. We repeat the same procedure for the total production vector ($\boldsymbol{X}^*$) and the final demand vector ($\boldsymbol{Y}^*$). The technical coefficient matrix in hybrid units ($\boldsymbol{A}^*$) can be calculated by:

$$\boldsymbol{A}^* = \boldsymbol{Z} * (\widehat{\boldsymbol{X}} *)^{-1} \tag{1}$$

---

[17] Energy generated by self-producers was not added since the majority of this energy is consumed by the same companies and therefore does not generate added value. Imported energy (which corresponded to approximately 7% of total energy supply in 2010) was also not included in our calculations since it was not possible to identify its respective sources.

The energy consumption in toe is then converted into the three main long- term drivers of climate change, CO_2, CH_4 and N_2O and later to CO_2e based on the energy conversion coefficient for fossil fuels[18] available from the Second Brazilian Inventory of Greenhouse Gas Emissions, which follows the IPCC Guidelines for National Greenhouse Gas Inventories (IPCC, 2007) and the global warming potential (GWP) conversion factors [19].

Assuming that CO_2e emissions by energy use are linearly related to the respective energy requirements, it is possible to estimate both direct emissions as well as total emissions for each good and economic activity. The matrix of technical coefficients of national inputs $\boldsymbol{A}_p$, is obtained by the product of $\boldsymbol{B}^*$ and $\boldsymbol{D}$. The matrix $\boldsymbol{B}^*$ contains, in hybrid units, the proportion of each domestic input used in the total production of a specific sector:

$$b_{ij} = \frac{u_{ij}^*}{\sum_i r_{ij}} \tag{2}$$

where u_{ij}^* is the element *ij* of the hybrid 'use' matrix, denoting the amount of domestic input *i* used in the production of sector *j*, and $\sum_i r_{ij}$ is the total production of sector *j*. Likewise, we calculate the share of product *i* produced by sector *j* (d_{ij}) as follows:

$$d_{ij} = \frac{r_{ij}}{\sum_j r_{ij}} \tag{3}$$

The coefficients of matrix $\boldsymbol{A}_p$ can be interpreted as the quantity of CO_2e that product *i* uses to produce one unit of product *j* (expressed in tonnes CO_2e/USD mi, in 2009 values [20] (Druckman and Jackson, 2009). Direct CO_2e emission coefficients are equivalent to the sum of the *k* rows of the $\boldsymbol{A}_p$ that measure emissions:

$$c_{i,CO_2e} = \sum_k ap_{kj} \tag{4}$$

in which $k \leq i$. We take the intermediate consumption coefficients and pre- multiply them by a Leontief inverse matrix:

$$c_{i,CO_2e}^T = c_{i,CO_2e} \cdot (I - \boldsymbol{A}_p)^{-1} \cdot Y \tag{5}$$

where Y is the vector of final demand. Therefore, the emission coefficient of the

---

[18] These conversion coefficients that take into account the characteristics of the chemical process and technology applied to each greenhouse gas. The following fuels were considered: natural gas, steam coal, metallurgical coal, diesel oil, fuel oil, gasoline, LPG, kerosene, gas coke, coal coke, other oil byproducts, and coal tar.

[19] The "global warming potential" (or GWP) of a GHG indicates the amount of warming a gas causes over a given period of time (normally 100 years). GWP is an index, with CO_2 having the index value of 1, and the GWP for all other GHG is the number of times more warming they cause compared to CO_2. E.g. 1 kg of methane causes 25 times more warming over a 100 year period compared to 1 kg of CO_2, and so methane has a GWP of 25.

[20] To reconcile with the Budgetary Household Survey, we converted 2010 values into 2009 values.

products corresponds to the total CO_2e content embedded in one monetary unit of the respective product.

Based on the premise of constant technological coefficients and returns to scale, the hybrid I-O model assumes no price substitution effect on consumption or pro- duction processes. It also considers that all interactions among components of the economy occur at the same time, instead of in a dynamic way. However, the premise of strict prices is adjusted by the estimation of a censored demand system, as detailed in Section 3.2. Therefore, emissions reductions are exclusively due to consumption changes and no supply-side effects are modelled with this approach (e.g. possible constraints to the supply of production factors - such as labor and capital - or technologies changes)[21]. In addition, our emission coefficient estimates include exports but exclude imports and do not take into account carbon leakages[22]. As our focus is on energy emissions from fossil fuels, we also do not consider the negative effects on the overall environment, such as on land use, soil erosion, fertilizer use[23].

Table A1 (Appendix) presents the estimates for the direct and indirect emission coefficient for the goods and services consumed by Brazilian households, presented in decreasing order of relative CO_2e emissions. We note very high coefficients for water and air transportation, as well as ground transportation of cargo and passengers, followed by wood products, cement, glass/ceramics, and food items such as dairy products, meat, beverages and canned foods. Total emissions tend to arise mainly from indirect emissions for most activities, which indicates there is higher energy consumption from the trade flows to meet the final demand for these respective goods/services. However, particularly for all types of transportation, wood products, paper and printing services, as well as selected agricultural and livestock products (such as rice, milk, cattle, poultry, eggs and fish), direct emissions might represent up to 90% of overall emissions. Therefore, these products/services exert significant influence on total GHG emissions in Brazil.

---

[21] In conventional IO model price changes are independent from quantity changes, so output changes mainly depend on changes in final demand (income effect). However, generally in CGE models, the production technology is modelled as a nested constant elasticity of substitution (CES) function in CGE models, and the Armington assumption is often applied - so similar inputs necessarily present different elasticities. In particular, energy can be substituted to the aggregate labor-capital input, and there is also possible substitution between energy types (e.g. electricity and petroleum products), which are endogenously determined. Therefore, when a CO_2 tax is applied as a VAT tax, this leads to a new optimization procedure to determine new price and quantity of the composite capital/labor-energy factors. For a detailed analysis of the modelling differences between CGE and conventional IO models, see Bun (2018).

[22] Carbon leakage refers to the phenomenon where overseas emissions (especially from those in countries with less strict environmental regulations) increase because of emissions restrictions in a given country

[23] Many studies have used CGE models to assess the impact of carbon tax on the economy. Differently from I-O models, a common practice in CGE modeling is using the social accounting matrix (SAM) as the main database to begin. The SAM is an account reflecting the circular transactions and transfers, in terms monetary value, between and within economic agents in an economy.

3.2 Households' responses to prices: censored QUAIDS

Carbon taxation mainly affects household expenditures on energy-related products and services like fuel and transportation, as these goods become more expensive. However, since carbon taxation also affects expenditures on non-energy related products by shifting the share of the household budget that is spent on each type of product, there is a need to understand the whole consumer behavior change caused by price modifications.

Consumer behavior theory says that individuals choose what and how much to consume to maximize their well-being, subject to a budget constraint. If the consumer's set of choices is consistent[24], the study of consumer behavior can be performed as a classic optimization problem[25], allowing the estimation of price and income elasticities.

However, consumer theory does not specify the functional forms for the demand equations. The advantage of estimating a system of demand equations instead of individual equations is based on the joint estimation and empirical tests concerning the validity of the theoretical restrictions implied in the consumer theory. We choose the Quadratic Almost Ideal Demand System (QUAIDS), which considers the nonlinearity of income, as presented below:

$$w_i = \alpha_i + \sum_{j=1}^{n} \gamma_{ij}\, ln(p_j) + \beta_i ln(\tfrac{m}{a(p)}) + \tfrac{\lambda_i}{b(p)}\, [ln(\tfrac{m}{a(p)})]^2 \tag{6}$$

where w_i is the expenditure for good i, p_j is the price of good n, m is the total expenditure per capita and ln $\alpha(p)$ is the transcendental price index, such that:

$$\ln[a(p)] = \alpha_0 + \sum_i^n \alpha_i\, ln(p_i) + 1/2\, \Sigma_i \Sigma_j \gamma_{ij} ln(p_i) ln(p_i) \tag{7}$$

and $b(p)$ is the Cobb-Douglas price aggregator, described as:

$$b(p) \ = \ \prod_{i=1}^{n} p_i^{\beta i} \tag{8}$$

and

$$\lambda(p) \ = \ \sum_i \lambda_i \ln p_i \tag{9}$$

The theoretical constraints on the model's parameters are:

$$\sum_{i=1}^{N} \alpha_i = 1; \sum_{i=1}^{N} \beta_i = 0; \sum_{i=1}^{N} \lambda_i = 0; \sum_{i=1}^{N} \gamma_{ij} = 0, \forall j \in I \tag{10}$$

---

[24] The consistency of preferences implies acceptance of the axioms of reflexivity, completeness, transitivity, continuity, no local satiety and strict convexity (Deaton and Muellbauer, 1980b).

[25] Due to consistency of consumer preferences, the system of demand equations has the properties of additivity, homogeneity, symmetry and negativity.

$$\sum_{j=1}^{N} \gamma_{ij} = 0, \forall i \in \tag{11}$$

$$\gamma_{ij} = \gamma_{ji}, \forall i \neq j \tag{12}$$

Following the demographic translation approach by Pollak and Wales (1981), we introduce sociodemographic shifters (z_j) by substituting (13) into (6) and (7). Demographic shifters are used to allow for household heterogeneity:

$$\alpha_i^* = \alpha_i + \sum_{j=1}^{n} \delta_{ij} z_j, \tag{13}$$

This procedure requires one additional constraint to the system of equations ($\sum_i^n \delta_{ij} = 0,\ \forall \mathrm{i} \in 1, \dots, \mathrm{n}$).

The empirical estimation of a demand system requires household expenditure data. We use data from the 2008-09 Brazilian Household Expenditure Survey[26], a nationally representative cross-sectional survey that contains data on all monetary and non-monetary household and individual expenses[27] during a given period, presented in different booklets. Food and beverage expenses are collected for a 7-day period; building material expenses, rent and taxes are compiled for a 12-month period; expenses related to the consumption of energy goods (electricity and fuels) are collected for a 90-day period; while individual expenses for transportation, education, meals outside the home, medicines, clothing and footwear, hygiene, health, furniture and vehicle acquisitions vary according to the good/service. The sample is based on a two-stage clustered sampling procedure, with a probabilistic selection of 550 household census sectors. Households within each sector are selected by simple random sampling without replacement and the interviews were carried out uniformly throughout the survey's four quarters to reproduce the seasonal variation in income and purchases in each stratum. For the purpose of this study, we use the household as unit of analysis.

We use the IBGE official translator[28] to reconcile almost 14,000 products available in the Household Expenditure Survey (POF) according to their similarity with the 98 products

---

[26] Microdata of 2017-2018 is available from April 10th of 2020. Unfortunately, the new microdata does not include quantity for most of the energy products (e.g. gasoline, ethanol and diesel) and it does not include expenditures with residential appliances (e.g. refrigerator, stoves, air conditioner, among others). Based on these shortcomings, it would not be possible to run the demand equations for the whole household consumption basket.

[27] The level of detailed information on monetary and non-monetary expenses and income from POF allows minimization of the under-declaration problem (Hoffmann, 2010).

[28] Available at: https://www.ibge.gov.br/estatisticas/economicas/contas-nacionais/9052-sistema-de-contas-nacionais-brasil.html?edicao=25916t=notas-tecnicas.

consumed by households in SUTs[29]. After this aggregation, we follow Ghalwash (2007) and Dorband et al. (2019) to group the combined categories of products into similar 9 main groups, which allows the understanding of the total household consumption: i) food and beverages, ii) recreation, culture and education, iii) clothing and footwear, iv) commuting and transportation, v) health and hygiene, vi) energy, vii) housing, viii) other goods and ix) other services. Table A2 provides a description of the items included in each of these main categories.

The use of household expenditure survey data for demand system estimation often creates a problem due to the lack of consumption of certain goods during the recall period. This causes censored dependent variables and leads to biased results when not accounted for. Following Shonkwiler and Yen (1999), the consumption of each good can be characterized as a two-stage decision: the first step corresponds to a probit model with the same variables as the QUAIDS model, in which its cumulative distribution ($\hat{\Phi}$) and the probability density function ($\hat{\varphi}$) are used in the second step to augment the QUAIDS estimation[30]:

$$w_i^* = \hat{\Phi}_i w_i + \hat{\phi}_i \tag{14}$$

The expenditure (15) and price elasticities (compensated, (16) and uncompensated, (17)) formulas for the non-linear QUAIDS can be expressed as:

$$\eta_i = 1 + \Phi_i / w_i [\beta_i + (\frac{2\lambda_i}{b(p)}) ln(\frac{m}{a(p)})] \tag{15}$$

$$\epsilon_{ij} = -\delta_{ij} + \Phi_i / w_i [\gamma_{ij} - (\beta_i + (\frac{2\lambda_i}{b(p)}) ln(\frac{m}{a(p)}))(\alpha_j + \sum_k \gamma_{jk} - lnp_k) - \frac{\lambda_i \beta_i}{b(p)} (ln(\frac{m}{a(p)}))^2 \tag{16}$$

where δ_{ij} is the Kronecker delta (equal to one only for own price elasticities, and zero otherwise):

$$\epsilon_{ij}^h = \epsilon_{ij} + (\frac{\beta_i}{w_i} + 1) w_j \tag{17}$$

To capture the heterogeneous effects of the energy tax policies, elasticities are calculated for the overall sample and among the 20% richest and 20% poorest households in the dataset. Income groups are constructed based on total household monetary and non-monetary income reported by POF. We use the information of total earnings as

---

[29] Some products available on SUTs are not consumed by households (e.g. pig iron and ferro-alloys).

[30] In the censored QUAIDS, the deterministic components on the right-hand side of equation set (14) do not add up to unity across all equations of the system in general, and so the error terms in the estimation form do not add to zero. Thus, the usual procedure of imposing the adding-up restriction (10) on the system and dropping one arbitrary equation is not valid. Therefore, with censoring, the second step of the system (14) is estimated correctly when using the entire set of equations (Yen, Kan, and Su, 2002).

stated in the POF, which contains wage, transfers, rental income, non- cash and other incomes to disaggregate the groups per different income levels. All models are estimated by feasible generalized non-linear Least squares (FGNLS), and standard errors are computed by nonparametric bootstrap with 1,000 repetitions. Since α0 is difficult to estimate (Deaton and Muellbauer, 1980b), we follow Boysen (2012) and adopt an arbitrary and low value of 5. Other values did not change the resulting elasticities but caused the procedure to require many more iterations to converge. Robustness checks are also conducted using uncensored QUAIDS and AIDS models with the STATA procedure suggested by Poi (2012) with the same specification.

This partial equilibrium framework assumes that the carbon tax burden is fully transferred to consumers. This is a reasonable assumption to be used in an assessment of the immediate impact given the structural stability in the short-term, in which the reallocation of factor input is unlikely (Grainger and Kolstad, 2010; Metcalf and Weisbach, 2009). In addition, the impact of a carbon tax on consumption in a demand system approach excludes the behavioral changes and possible welfare benefit received from reduced emissions. Due to these limitations, the estimates can be interpreted as the carbon tax policy’s upper bound effect.

Table A3 (Appendix) shows the descriptive statistics for positive consumption and budget shares for each group and income level. For instance, the household bud- get share allocated to food and beverage and energy consumption among the 20% poorest is more than double the proportion of spending on these products among the 20% richest households. For commuting and transportation services, as well as for recreation and education, this proportion is even higher. However, the richest and poorest households tend to spend similar shares on clothing and footwear and housing. Furthermore, on average, censoring is higher for health and hygiene items, as well as for culture and private education, mainly because poorer households have lower consumption of goods from these groups. Clothing and footwear expenses are presented in different booklets, which explains the high percentage of positive consumption for all income levels. Overall, these figures justify the use of the censored approach when using data from household expenditure surveys.

The descriptive statistics of socioeconomic variables is presented in Table A4 (Appendix), to help explain the differences in preferences of households for the products analyzed. Heads of the 20% richest households have almost 5 more years of education compared to the heads of the 20% poorest households. In addition, the richest group has more than twice as many rooms and bathrooms in their houses (good proxy for wealth) compared to the poorest group, on average. Total per capita earnings of high-income households are 15 times higher than for low-income households.

3.2.1 Construction of prices

The main theoretical variables for household demand system are, basically, total expenditures (proxy for income)[31] and prices, calculated as unit values ($p_i = UV_i$). Particularly for the products from group 1 (food and beverages), there are two main problems related to the price we calculate from the household expenditure surveys: potential measurement error, and differences in quality and packaging (Boysen, 2012). In this sense, we use a price correction method from Cox and Wohlgenant (1986) and Lazaridis (2003) for these specific products.

As not all household have positive consumption of all items, the missing observations are approximated by the average of $\hat{p_i}$ coefficients over the neighboring region - first, the stratum and if it is still missing, the state. Then we compute the weighted price indexes[32] (Stone price index) for all groups (Deaton and Muellbauer, 1980a):

$$lnp_g = \sum_{i\epsilon I_g} w_i lnp_i \tag{18}$$

in which I_g is the set of items included in aggregate item group g, p_i is the price and w_i is the budget share of item i in each household. Because expenditures and prices are endogenous in this demand system, we use households' disposable income as an instrument for expenditures, and nearest neighbors' price indexes as instruments for household price indexes (Lecocq and Robin, 2015), controlling for diversity in household preferences such as their composition, age and geographical location.

In the POF, there is a limitation related to the lack of specification of the quantity consumed of several products and services: in a 12-month period (e.g., rent, taxes, construction and remodeling) and on an individual basis (e.g., education, commuting and transportation). This affects items aggregated into groups 2, 3, 4, 5, 7 and 9. We assume that the quantity consumed was equal to 1 for the households with positive consumption of the respective product or service. This approach tends to overestimate the unit value of some products and services, thus underestimating the respective price elasticities.

Based on these limitations, Table A5 (Appendix) presents the aggregated prices, in which values expressed in Brazilian reais of 2009 are converted to 2019 using the average exchange rate from the Central Bank of Brazil for that year. The price indexes should be interpreted as a relative price index: for example, richer households expend 34%, 78% and 67% more on food and beverages, commuting and transportation services and energy goods, respectively, compared to poorer households. The small standard deviation values are associated with the lack of information related to quantity consumed of several

---

[31] Since household income is self-reported, this information might be associated with negative reporting bias. To overcome this issue, the literature usually adopts household total expenditure as a proxy for household income.

[32] For items of group 1, this procedure is implemented after the price correction.

expenditure items.

3.3 Calculation of effects

Welfare and emissions effects, as well as the rationale behind revenue neutrality, are explained below. These effects are calculated for the average-income of Brazilian households as well as for the 20% richest and 20% poorest households, and the estimations are multiplied by the number of Brazilian households from the 2010 Brazil Demographic Census (approximately 57 million inhabitants) to calculate these effects for the country as a whole. Since the impacts calculated in this study are mainly valid for the short-run, they should be interpreted as upper bounds for long-term impacts.

3.3.1 Welfare effects

Assuming that prices are fully transferred to consumers and focusing only on the costs associated with the tax imposition, we use the concept of equivalent variation (EV) to assess the short-term effects of a carbon tax on welfare. The EV, expressed in monetary terms, indicates the maximum amount the consumer would be willing to pay to avoid a price change caused by the introduction of a tax:

$$EV = e\ (p^1, u^1) - e\ (p^0, u^1) \tag{19}$$

where e is the expenditure function considering the ex-post utility (u^1) at pre- tax and post-tax prices, respectively. Likewise, in a situation in which the taxpayer cannot take any action to influence the amount of taxes paid (tax evasion), the dollar magnitude of the welfare loss as measured by the EV will exceed the total tax revenue collected from the taxpayer – and the difference is defined as the deadweight loss of the tax or the excess tax burden. Therefore, the deadweight loss (DWL) (corresponding to the EV) represents the efficiency loss arising from the tax - that is, utility that is lost beyond the revenue transferred to the government (Mohring, 1971):

$$DWL\ (u^1) = EV - (p^1 - p^o)\ h\ (p^1, u^1) \tag{20}$$

where $h(.)$ is the compensated demand function. One virtue of an equivalent variation measure of excess burden lies in the fact that in comparing tax policies that raise equal revenue, the tax policy with the lowest excess burden as measured by equivalent variation also produces the highest level of consumer welfare (Kay, 1980).

A key issue to examine the distributional effects of carbon taxes is how to measure the magnitude of tax burdens between poor and rich households. Many households in the lower income deciles dis-save on previous earnings or may borrow against future earnings. Their level of expenditure reflects better what they are able to afford than their level of income (Cronin, Fullerton, and Sexton, 2019; Flues and Thomas, 2015). Since current consumption measures the current standard of living better than current income, we present both EV and DWL estimates relative to the total household expenditures.

3.3.2 Emissions effects

The difference between total GHG emissions before and after the tariff rate changes indicates the changes in total household carbon footprints due to the carbon tax policy, as follows:

$$\Delta CO_2e = \sum\nolimits_g (p_g^1 * q_g^1) * c_{g,CO_2e}^T - \sum\nolimits_g (p_g^0 * q_g^0) * c_{g,CO_2e}^T \tag{21}$$

3.3.3 Neutral revenue

We estimate the additional tax revenue collected by the government considering the tax rate, the total household expenditure and the carbon intensity of each group consumed. As per the “polluter pays” principle and the double-dividend rationale, we consider that the government revenue obtained from the carbon tax is fully redistributed to the households as a lump-sum transfer.

3.4 Scenarios and sensitivity analysis

Since tax rates implemented vary significantly worldwide, we simulate two scenarios considering USD 40/tCO_2e and USD 80/tCO_2e. According to the High-Level Commission on Carbon Prices - a group of leading economists working with the Carbon Pricing Leadership Coalition -, the explicit carbon-price level consistent with achieving the Paris temperature target is at least USD 40–80/tCO_2 by 2020 and USD 50–100/tCO_2 by 2030. These estimates are also aligned with the prices calculated by the US Interagency Working Group on the Social Cost of Carbon of USD 50/tCO_2 in 2020. The International Monetary Fund (IMF, 2019) also estimates that prices from USD 50/tCO_2 to USD 100/tCO_2 or more by 2030 are needed to meet their commitments to reduce carbon emissions.

The rate is applied to CO_2, CH_4 and N_2O emissions from fossil fuels to Brazil’s 2009 productive structure and USD values for 2019. We also conduct sensitivity analysis by: i) narrowing the tax base according to the groups of products and services that present the highest carbon content - food and beverages and commuting and transportation services, respectively, and ii) applying a social tariffs on energy goods (group 6) consumed by the poorest households using carbon tax revenues from richest households. In particular, the expansion of direct transfers to poorest households is current part of one package proposal being discussed under the context of the Brazilian tax reform; however, this proposal has received a lot of criticism as it does not identify potential sources of revenues to be transferred in the form of direct transfers. Therefore, in this scenario, we assess the extent of which the distributional effect of a carbon tax could be minimized by directly using the revenues from a carbon tax paid by richest households.

4 Results and Discussions

4.1 CO_2e emission coefficients

Table A6 (Appendix) presents the estimates for total CO_2e emissions per household - or carbon footprint -, obtained by multiplying total expenditures (at 2019 values) and the emission coefficient for each group of products. Food and beverages and commuting and transportation have the highest averages of CO_2e emissions per household (3.3 tCO_2e/hh/year and 2.0 tCO_2e/hh/year, respectively). Commuting and transportation mainly refers to cargo transportation, which is predominantly done over highway networks using diesel as fuel, and individual transportation, with prevalence of flex-fuel cars (ethanol and/or gasoline in any combination). It also accounts for public passenger transportation, largely by buses. Therefore, the high levels of CO_2e emissions from the food and beverages group is a reflection of the country's characteristics of cargo transportation and the large expenditure share devoted to this group. The relatively low levels of CO_2e emissions from the domestic energy group is explained by the fact that, on average, the majority of household expenditures refer to electricity and gasoline consumption. Despite an increasing share of production coming from natural gas and coal, electricity is still primarily generated by hydropower sources.

Other studies have found that carbon emissions embedded in transportation and commuting services, as well as food items, generally form a substantial portion of a household's carbon footprint, in particular when analyzed in terms of greenhouse gas emissions instead of carbon dioxide only. Differently from what we observe for Brazil, empirical studies of Australia (Dey et al., 2007), Netherlands (Nijdam et al., 2005) and UK (Druckman and Jackson, 2009) also point out that an important share of GHG emissions also arise from expenses related to heating, electricity and house maintenance, normally aggregated in the housing group.

Regarding the distribution of emissions among income levels, we note much higher figures for the richest 20% of Brazilian households, given that the total annual emissions in this category reach 26 tCO_2e per household, far above the total calculated for the 20% poorest (1.7 tCO_2e). Indeed, many studies have shown that the relationship between income and household carbon footprint is strong (Wier et al., 2005; Dey et al., 2007; Perobelli, Faria, and Almeida Vale, 2015). Our estimates suggest that 25% of total emissions from low-income households arise from food and beverage consumption, while 30% of GHG emissions from high-income households come from commuting and transportation services. Therefore, since expenditures on commuting and transportation increases with income, and this group has one of the highest emission intensity coefficients, these two coupled components play an important role in terms of reducing emissions. Likewise, households with lower income level tend to have a structure of spending that more intensively mobilizes the inputs related to the food production chain - such as transportation.

Our carbon footprint results also show adherence to the metrics disclosed by the World Bank[33]for the Brazilian economy. According to its estimates, the average emission per household in Brazil was approximately 9.9 tCO_2 in 2014, roughly in line with the average of 10.5 tCO_2 resulting from our approach. Furthermore, taking 2014 World Bank statistics as a benchmark, we find that the average emission per household in Brazil is significantly lower than the global average. Indeed, the average CO_2e emission of the richest quintile in Brazilian population is below the average emission considering only CO_2 in China (30 tCO_2/household), Germany (35.6 tCO_2/household) and United States (66 tCO_2/household). Likewise, the average CO_2e emission of the poorest quintile is much lower than the average CO2 emission in India (6.8 tCO_2/household).

4.2 Expenditure, own- and cross- price elasticities

Table A7 (Appendix) presents the expenditure, own and cross-price elasticities of household groups, obtained through the demand system estimation. These elasticities measure the effectiveness of a pricing policy, such as a carbon tax, and determine the vulnerability of households in reducing their energy consumption when energy prices increase as a result of a carbon tax.

Expenditure elasticities for food items are low (0.8), especially for the 20% poorest households (0.5). This suggests that, compared to other groups, food de- mand is much less responsive to changes in income. On average, the expenditure elasticities for domestic energy goods are relatively high (1.8), indicating that they are luxury goods for all levels of income - especially for low-income households (1.9).

In contrast, expenditure elasticities for housing (which contains rent, as well as residential appliances) are relatively low (0.5) - particularly for high-income groups (0.2). Interestingly, for richer households, commuting and transportation services are considered necessity goods (0.9), while for poorer households they can be classified as superior goods (1.9). The demand for recreation and cultural activities from high-income households is less responsive to changes in income (0.6) compared to poorer households (3.9), being characterized as necessity goods and luxury goods for richer and poorer households, respectively.

---

[33] Data for carbon dioxide emissions include gases from the burning of fossil fuels and cement manufacture, but exclude emissions from land use such as deforestation. The U.S. Department of Energy's Carbon Dioxide Information Analysis Center (CDIAC) calculates annual anthropogenic emissions from data on fossil fuel consumption (from the United Nations Statistics Division's World Energy Data Set) and world cement manufacturing (from the U.S. Department of Interior's Geo- logical Survey, USGS 2011). Estimates exclude fuels supplied to ships and aircraft in international transport because of the difficulty of apportioning the fuels among benefiting countries. Although estimates of global carbon dioxide emissions are probably accurate within 10 % (as calculated from global average fuel chemistry and use), country estimates may have larger error bounds. Each year the CDIAC recalculates the entire time series since 1949, incorporating recent findings and corrections.

One interpretation of these empirical results is that, in general, food, housing and other goods are seen as urgent needs, and thus as budgets increase, these services tend to be prioritized. Most goods and services are luxuries relative to food, housing and other goods - as income rise, the willingness to spend more on these goods and services increase more than proportionally. Saturation effects[34] are also observed for energy, commuting and transportation, recreation and education as well as other services, meaning that richer households demand proportionally less goods and services from these groups but more of other goods and services. This suggests that consumption of and expenditure on many items previously considered to be luxury goods and services would grow less than income.

Studies estimating elasticities of several household expenditure groups using similar demand system approach have found income elasticities for food items ranging from 0.4 for Germany (Nikodinoska and Schröder, 2016) up to 0.7 for Italy (Tiezzi, 2005) and 0.8 for Sweden (Brännlund and Nordström, 2004). Our estimations related to commuting and transportation are similar to the ones found for Italy (Tiezzi, 2005) and Spain (Labandeira, Labeaga, and Rodríguez, 2006) for public transport. For Sweden, Ghalwash (2007) identified total expenditure elasticities of ranging from 0.2 to 0.4 for food and beverages, 2.1 for recreation, 1.1 for clothing, 0.5 for transports, 0.9 for health care and from 0.3 to 1.2 for energy goods. Also for Germany, Reaños and Wölfing (2018) found that expenditure elasticities for energy goods are typically smaller in absolute value among more affluent households.

Uncompensated (Marshallian) and compensated (Hicksian) own-price elasticities show the expected negative signs. On average, household demand is inelastic with respect to the consumption groups with high embodied carbon content. Low- income households are less price-responsive for the majority of carbon-intensive categories (food and beverages (-0.8), commuting and transportation (-0.4), energy (-0.8), housing (-0.8) and other goods (-0.3)). For these categories, richer households present elasticities of -1.0 (Food and Beverages), -1.1 (commuting and transportation, energy, housing and other goods). Therefore, poorer households can be expected to reduce their consumption less than rich households due to tax-induced price increases in these categories, while the reduction in health and hygiene (-1.6), recreation and education (-1.4), clothing and footwear (-1.5) and other services (-1.3) is relatively higher. Considering responses in demand, the real expenditure loss would, therefore, be higher for poorer households, which would make distributional effects more regressive.

Our findings are aligned with Labandeira, Labeaga, and López-Otero (2017) and

---

[34] Saturation effects imply that beyond a certain level of consumption, there is a declining share of the budget allocated to certain goods and services as incomes rise – and thus income elasticities for those good and services also fall.

Ghalwash (2007) and Dorband et al. (2019). In particular, Dorband et al. (2019) assessed the expected incidence of moderate carbon price increases for different income groups in 87 mostly low- and middle-income countries and found own-price elasticities ranging from -0.4 to -0.7 for food, beverages and tobacco, -0.7 for clothing and footwear and -0.7 for education.

The compensated own-price elasticities (used in the calculation of welfare effects) indicate similar patterns but in lower absolute terms since only the substitution effect is included and expenditure elasticities are all positive (Tables A8, Appendix). Cross-price elasticities (Table A9, Appendix) suggest that, in general, the groups with the highest CO_2e emission coefficient (commuting and transportation and other goods) are substitutes for food items. Food and beverages are also substitutes for housing and commuting and transportation services. Commuting/transportation is also a substitute for housing and energy. However, energy goods appear to be complementary for housing.

In general, the findings suggest larger behavioral adjustments for rich house- holds when facing changes in prices of carbon-intensive goods and services - such as food, commuting and transportation and recreation - while low-income households are already required to focus on their basic needs. Thus, high-income households should generate more emissions and a small variation in the consumption could have a significant impact on emissions. In addition, changes in the quantity demanded of food items appear to have a significant impact on the consumption of other carbon- intensive goods and services, and therefore influence the overall CO_2e emissions.

4.3 Welfare and emissions effects

Table A10 (Appendix) presents our analysis of the distributional implications of an economy-wide carbon tax of USD 40/tCO_2e and USD 80/tCO_2e on welfare and emissions, in absolute terms and as a percentage of total household expenditure. The results indicate that the first-dividend effect is observed and the taxation policy is Capable of achieving its main goal: to reduce emissions. On average, a carbon tax of USD 40/tCO_2e would be able to reduce overall household emission by 2.0% per year (equivalent to a reduction of 12.5 $MtCO_2e$), reaching approximately 4.2% (25.2 $MtCO_2e$) considering a tax of USD 80/tCO_2e. High-income households are mainly responsible for the largest part of the reduction of total emissions, accounting for decreases of 7.6 $MtCO_2e$ (-2.5%) and 14.6 $MtCO_2e$ (-4.9%) per year, respectively. This pattern of emission reduction has been largely observed in many other empirical studies of developed countries (Grainger and Kolstad, 2010; Fremstad and Paul, 2019) as well as of other developing countries (Renner, 2018; Brenner, Riddle, and Boyce, 2007), including Brazil (Magalhães and Domingues, 2013; Silva Freitas et al., 2016).

Regarding the welfare impact associated with the carbon tax (measured as equivalent variation in monetary equivalent of the change in utility), we observe aggregate welfare losses of USD 237 mi and USD 244 mi for a carbon tax of 40/tCO_2e and 80/tCO_2e respectively, approximately 0.02% of total household expenditure, respectively. The taxes raise the price of more carbon-intensive products and reduce CO_2e emissions by their negative impact on consumption, thus lowering welfare in all cases - since the social, economic and environmental co-benefits of reducing CO_2e emissions are not taken into account in this analysis, which might benefit poorer households more than the richer households (Gao et al., 2018a; Gao et al., 2018b)[35].

We also identify the disproportionality of the welfare losses induced by the carbon tax across different types of households. The higher share of EV as a percentage of expenditure for poorest households (0.10%) vis-a-vis richer households (0.06%) suggests that the policy is regressive. Results also indicate that even at the higher tax rate considered (of USD 80/tCO2e), the carbon tax has very little incremental impact on the behavior of households[36]. Using similar tax rates, welfare losses of higher magnitude were found for low-income households in Sweden (0.52%; Brannlund and Nordström (2004)), Denmark (0.8%; Wier et al. (2005)), U.S. (3.7%; Grainger and Kolstad (2010) and Hassett, Mathur, and Metcalf (2007)), Mexico (4.2%; Renner (2018)), Italy (0.4%; Tiezzi (2005)) and Brazil (3.1%; Silva Freitas et al. (2016)) and 2.2%; Magalhães and Domingues (2013)). When compared to studies which uses CGE models, our results tend to be relatively overestimated, as I-O models do not allow for supply side effects (Metcalf and Weisbach, 2009).

However, due to the consumption patterns, the deadweight loss estimates caused by the carbon tax show that richer households would have an additional tax burden of 0.04% over the total expenditure per year (USD 205 million and USD 209 mil- lion for a USD

---

[35] A range of carbon pricing co-benefits have been identified in the literature, and several studies have attempted to quantify these in economic terms, also looking at its distributional effects. Co- benefits include improved public health from a reduction in local air pollution, increased energy security, and positive outcomes for land-use and biodiversity from the forest and agriculture sectors. The health co-benefits of carbon pricing have received the most attention. Studies indicate that developing countries would have more to gain from climate policy health co-benefits, in particular India, China and other East-Asian countries (Nemet, Holloway, and Meier, 2010). However, despite overall projected gains, conditions could also worsen in the short term for some households in areas of South-Asia and Africa where climate policies could drive greater use of biomass among poor households for cooking and heating (West et al., 2013; Rao et al., 2016). Likewise, as health effects from co-pollutants depend on their local concentrations, communities living nearest to the sources are generally assumed to have the most to gain from local climate action. For example, a study of pollution patterns across the U.S. found that racial minorities, low-income families and elderly people were more likely to be exposed to higher levels of pollution due to where they live (Clark, Millet, and Marshall, 2014). A worldwide literature review found similar patterns across North America, Asia and Africa, but found mixed results across Europe (Hajat, Hsia, and O'Neill, 2015). For Brazil, Guidetti, Pereda, and Severnini (2020) show that pollution reduction from transportation services in the metropolitan region of Sao Paulo reduces the number of hospital admissions from respiratory-diseases, potentially for lower-income population.

[36] It is worth mentioning that lump sum transfers might not reach the most vulnerable groups within the poorest households. As an example, poor rural communities of the Amazon riverside (also known as "*ribeirinhos*"), still suffer from the lack of electricity benefits, such as lighting and refrigeration, and normally pay relatively higher prices for gas and oil for their subsistence activities. It is most likely that only households already enrolled in other cash transfer programmes -such as *Bolsa Família* - would receive any redistribution. When accounting for this aspect, the distributional effects of an economy-wide carbon tax could be even more regressive.

40/tCO_2e and USD 80/tCO_2e tax, respectively), compared to 0.02% faced by poorer households.

The results also indicate that with a tax rates of USD 40/tCO_2e and USD 80/tCO_2e, and a relatively broad tax base - levied across all goods and services in the Brazilian economy based on its carbon content -, the government could increase its revenues by USD 616 million and USD 630 million per year, respectively, which would be equivalent to approximately 0.05% of total federal tax revenue. As a comparison, the total federal revenue obtained in 2018 from CIDE-fuels was USD 16.7 billion, which corresponds to 0.23% of total federal tax revenue.

To avoid an increase of the burden on taxpayers, if every dollar is returned to Brazilian households in a lump-sum transfer, high-income households would receive USD 103 million - approximately 0.02% of their total expenditure-, while low-income households would receive USD 64 million - equivalent to 0.7% of their total expenditure, respectively. Therefore, due to the regressive effects of the policy, a carbon tax scheme should be followed by a compensation policy, such as a direct transfer, which could offset the negative impacts of the tax on poorer households. Given the complexity of the Brazilian tax system - with over 80 different taxes and other fiscal levies at the federal, state and municipal levels -, together with the high tax burden 33% of gross domestic product (GDP), the second dividend effect could occur in a context in which the carbon tax is implemented under a broader tax reform.

4.4 Sensitivity analysis

If a carbon tax is implemented as a discretionary policy, levied only on groups of products with high carbon footprint - such as commuting and transportation and food and beverages -, the environmental impact would be positive, but much smaller in magnitude, as presented in Tables A11 and A12 (Appendix).

Following this design, total carbon emissions are expected to decline by 0.4% and 0.9% with a carbon tax of USD 40/tCO_2e, and 0.9% and 1.7% for USD 80/tCO_2e tax if it is levied on commuting and transportation and on food and beverage goods and services, respectively. Despite representing a small change of the share of total household expenditures, the economic inefficiencies, as well as welfare losses, would be relatively higher in magnitude in the scenario in which only food and beverages are taxed based on their carbon content, especially for poorer households. In order to preserve revenue-neutrality, the government should transfer approximately USD 63 million and USD 1.8 million to high- and low-income households, respectively, if a hypothetical carbon tax is levied only on commuting and transportation services, and USD 70 million and USD 3.3 million for a carbon tax applied to food and beverages, respectively. These results suggest that implementing a carbon tax for targeted products, based on their carbon content, is less regressive than an economy- wide carbon tax.

Likewise, Table A13 (Appendix) presents the results of the simulation in which carbon tax revenues from richest households are directly transferred to the poorest households as a social tariff, used exclusively for the consumption of energy goods (group 6). As approximately 70% of energy goods' expenditures are related to electricity and gas consumption (see Table A14), we calculate that this policy would soften the overall regressiveness of the carbon since the additional tax burden is only 1/4 of the estimated in the economy-wide carbon tax scenario. However, as energy goods are one important source of carbon emissions among Brazilian households, emission reductions would also be significantly smaller when compared to a broader carbon taxation.

5 Conclusions and Policy Implications

As part of the Paris Agreement, Brazil assumed, through its NDC, a commitment to reduce GHG emissions by 37% below 2005 levels in 2025 and subsequently by 43% below 2005 levels in 2030. We analyze the effectiveness of implementing an economy-wide carbon tax as an option among carbon pricing mechanisms, given that a tax system reform is a top-priority for the current Brazilian government and Brazil's NDC should be revised in 2020 - document that is not clear about which instruments might be adopted by the country to reach its 2030 goals.

Notwithstanding the attractiveness of a carbon tax policy to possibly sustain mid-term environmental targets, potential distributional issues are relevant from the normative perspective since they can affect the acceptability of the policy and put a question mark on its overall effectiveness. Within a partial equilibrium framework, our analysis offers a detailed assessment of the distributional short-term welfare and emission effects of a hypothetical carbon tax in Brazil.

Our findings suggest that the first dividend effect could be observed in the Brazilian context: at the benchmark levels of USD 40/tCO_2e and USD 80/tCO_2e, a carbon tax can provide an improvement in the environmental conditions, as it reduces overall GHG emissions by 2.1% and 4.2%, respectively. However, besides being insufficient to comply with the Paris Agreements, evidence indicates that a carbon tax tends to be regressive by causing welfare losses of 0.06% and 0.10% in relation to total expenditures for richest and poorest households, respectively. Low-income households are less price-responsive for the majority of carbon intensive categories, so they suffer a larger relative welfare loss due to the carbon tax. They are also more likely to suffer from a larger relative indirect effect of food and beverages and housing-related consumption, which accounts for a greater budget share of these households. Significant changes in total GHG emissions would require a higher tax rate, which would reinforce the regressiveness of the policy.

The sensitivity analysis also shows that implementing a discretionary carbon tax policy by narrowing the tax base and focusing only on products with high carbon content (commuting and transportation and food and beverages) could reduce the regressiveness of the carbon tax. The same can be observed if a social tariff to poorest households is financed by the carbon tax's revenues from richest households. Despite that, the reductions in overall emissions would be equivalent to 22%, 41% and 39%, respectively, of the total reduction in GHG emissions with an economy-wide car- bon tax, which makes more difficult the achievement of international environmental commitments.

The findings indicate that compensation strategies, such a direct lump-sum transfer, need to be considered by the government to reduce the burden imposed on these households. Brazil already has one of the heaviest tax burden among developing countries (around 33% of the GDP), which is close to the average of the countries comprising the Organization for

Economic Cooperation and Development (OECD). Unlike developed economies, however, the Brazilian burden is more concentrated in indirect and regressive taxes, as opposed to direct and progressive ones. Implementing a carbon tax within the current regulatory framework, which already generates distortions, would worsen the overall regressivity of the Brazilian tax system. In this sense, the generation of the second dividend effect could be observed only if the country implements this carbon pricing mechanism as part of a broader structural tax reform, following examples such as Argentina, Mexico and Colombia.

With this taken into account, there would be additional "green" opportunities that, in conjunction with a market-based measure (such as the carbon tax) might be able to boost GHG emissions reduction in the country. Given the aim of the country to reach 23% of non-hydro renewables in its power mix by 2030 (BRASIL, 2015), as well as the adjustments in the regulatory framework (such as the 2017 National Policy for Biofuels, the RenovaBio, approved by the Federal Law 13.576), carbon revenues could be used to fund investments and projects in solar, wind and biomass. In addition, investments in the stalled rural electrification program could be resumed, as Brazil still has people with no access to electricity - specially in isolated areas in the Amazon states, where 10% of the households do not have access to electricity (Silveira Bezerra et al., 2017)

A few caveats deserve attention, which could be explored by future studies. First, we assume that the changes in energy prices from a carbon tax are fully passed through to consumers with no supply-side effect. Carbon pricing may change real wages and returns to capital, which can influence the optimal input production (and hence emissions) for various sectors. Second, our analysis focuses only on the cost of the policy; and direct burden is only one channel through which a climate policy has distributional effects. Empirical evidence suggest that low-income households can obtain more gains from co-benefits of the carbon tax, so the 'net' incidence of the policy could actually be progressive (Gao et al., 2018a; Gao et al., 2018b). Third, despite considering an economy-wide carbon tax, one which takes into ac- count all emissions, actual carbon pricing mechanisms often have exemptions for emissions from some industries due to political considerations or high monitoring costs. Fourth, because we do not observe prices and quantities for all products included in the Brazilian household expenditure survey, our short-term elasticity estimates might be underestimated for several consumption groups. A suggestion for the next POF for IBGE to disclose the prices of the products used in the calculation of the consumer price index for the period in which the survey is carried out. This would improve the accuracy of future studies investigating consumer behavior patterns.

References

Aidy, Joseph E (2017). *The Political Economy of Carbon Pricing Policy Design*. Harvard Project on Climate Agreements.

Babiker, Mustafa H, Metcalf, Gilbert E, and Reilly, John (2003). "Tax distortions and global climate policy". In: *Journal of Environmental Economics and Management* 46.2, pp. 269–287.

Beck, Marisa et al. (2015). "Carbon tax and revenue recycling: Impacts on households in British Columbia". In: *Resource and Energy Economics* 41, pp. 40–69.

Bowen, Alex (2015). *Carbon pricing: How best to use the revenue.* Policy Brief. Grantham Research Institute and Global Green Growth Institute.

Boyce, James K (2016). "Distributional issues in climate policy: air quality co- benefits and carbon rent". In: *Amherst, MA: University of Massachusetts, Political Economy Research Institute, Working Paper* 412.

— (2018). "Carbon pricing: effectiveness and equity". In: *Ecological Economics* 150, pp. 52–61.

Boysen, Ole et al. (2012). *A food demand system estimation for Uganda*. Tech. rep. IIIS.

Brannlund, Runar and Nordström, Jonas (2004). "Carbon tax simulations using a household demand model". In: *European Economic Review* 48.1, pp. 211–233.

BRASIL (2015). *Pretendida Contribuição Nacionalmente Determinada (Indc) Para Consecução do Objetivo da Convenção-Quadro das Nações Unidas sobre Mudança do Clima*. url: http://www.itamaraty.gov.br/images/ed_desenvsust/ BRASIL-iNDC-portugues.pdf.

— (2019). *Sistema de Estimativas de Emissões de Gases de Efeito Estuda (SEEG).*

Brenner, Mark, Riddle, Matthew, and Boyce, James K (2007). "A Chinese sky trust? Distributional impacts of carbon charges and revenue recycling in China". In: *Energy Policy* 35.3, pp. 1771–1784.

Bureau, Benjamin (2011). "Distributional effects of a carbon tax on car fuels in France". In: *Energy Economics* 33.1, pp. 121–130.

Callan, Tim et al. (2009). "The distributional implications of a carbon tax in Ireland". In: *Energy Policy* 37.2, pp. 407–412.

Carattini, Stefano, Carvalho, Maria, and Fankhauser, Sam (2018). "Overcoming public resistance to carbon taxes". In: *Wiley Interdisciplinary Reviews: Climate Change* 9.5, e531.

Clark, Lara P, Millet, Dylan B, and Marshall, Julian D (2014). "National patterns in environmental injustice and inequality: outdoor NO 2 air pollution in the United States". In: *PloS one* 9.4, e94431.

Cox, Thomas L and Wohlgenant, Michael K (1986). “Prices and quality effects in cross-sectional demand analysis”. In: *American Journal of Agricultural Eco- nomics* 68.4, pp. 908–919.

Cronin, Julie Anne, Fullerton, Don, and Sexton, Steven (2019). “Vertical and horizontal redistributions from a carbon tax and rebate”. In: *Journal of the Association of Environmental and Resource Economists* 6.S1, pp. 169–208.

Deaton, Angus (1990). “Price elasticities from survey data: extensions and Indonesian results”. In: *Journal of econometrics* 44.3, pp. 281–309.

Deaton, Angus and Muellbauer, John (1980a). “An almost ideal demand system”. In: *The American economic review* 70.3, pp. 312–326.

— (1980b). *Economics and consumer behavior*. Cambridge University Press.

Devarajan, Shantayanan et al. (2011). “Tax policy to reduce carbon emissions in a distorted economy: Illustrations from a South Africa CGE model”. In: *The BE Journal of Economic Analysis & Policy* 11.1.

Dey, Christopher et al. (2007). “Household environmental pressure from consumption: an Australian environmental atlas”. In: Sydney University Press.

Dorband, Ira Irina et al. (2019). “Poverty and distributional effects of carbon pricing in low- and middle-income countries–A global comparative analysis”. In: *World Development* 115, pp. 246–257.

Druckman, Angela and Jackson, Tim (2009). “The carbon footprint of UK house- holds 1990–2004: a socio-economically disaggregated, quasi-multi-regional input– output model”. In: *Ecological economics* 68.7, pp. 2066–2077.

Edenhofer, Ottmar (2015). *Climate change 2014: mitigation of climate change*. Vol. 3. Cambridge University Press.

EPE, Empresa de Planejamento Energético (2018). “Balanco Energético Nacional, 2019”. In: *Relatório Síntese, ano base*.

Feng, Kuishuang et al. (2010). *Distributional effects of climate change taxation: the case of the UK*.

Flues, Florens and Thomas, Alastair (2015). *The distributional effects of energy taxes*. Tech. rep.

Fremstad, Anders and Paul, Mark (2019). “The Impact of a Carbon Tax on Inequality”. In: *Ecological Economics* 163, pp. 88–97.

Fullerton, Don (2011). “Six distributional effects of environmental policy”. In: *Risk Analysis: An International Journal* 31.6, pp. 923–929.

Fullerton, Don and Heutel, Garth (2011). “Analytical general equilibrium effects of energy policy on output and factor prices”. In: *The BE Journal of Economic Analysis & Policy* 10.2.

Fullerton, Don and Metcalf, Gilbert E (1997). *Environmental taxes and the double- dividend hypothesis: did you really expect something for nothing?* Tech. rep. National bureau of economic research.

Fullerton, Don and Muehlegger, Erich (2019). "Who bears the Economic Burdens of environmental regulations?" In: *Review of Environmental Economics and Policy* 13.1, pp. 62–82.

Gao, Jinghong et al. (2018a). "Greenhouse gas emissions reduction in different eco- nomic sectors: Mitigation measures, health co-benefits, knowledge gaps, and pol- icy implications". In: *Environmental pollution* 240, pp. 683–698.

Gao, Jinghong et al. (2018b). "Public health co-benefits of greenhouse gas emissions reduction: A systematic review". In: *Science of the Total Environment* 627, pp. 388–402.

Ghalwash, Tarek (2007). "Energy taxes as a signaling device: An empirical analysis of consumer preferences". In: *Energy Policy* 35.1, pp. 29–38.

Goulder, Lawrence H (1995). "Environmental taxation and the double dividend: a reader's guide". In: *International tax and public finance* 2.2, pp. 157–183.

Grainger, Corbett A and Kolstad, Charles D (2010). "Who pays a price on carbon?" In: *Environmental and Resource Economics* 46.3, pp. 359–376.

Grottera, Carolina, Pereira Jr, Amaro Olimpio, and La Rovere, Emilio Lèbre (2017). "Impacts of carbon pricing on income inequality in Brazil". In: *Climate and Development* 9.1, pp. 80–93.

Guidetti, Bruna, Pereda, Paula, and Severnini, Edson (2020). "Health Shocks under Hospital Capacity Constraint: Evidence from Air Pollution in Sao Paulo, Brazil".

Guilhoto, Joaquim et al. (2010). "Estimação da Matriz Insumo-Produto Utilizando Dados Preliminares das Contas Nacionais: Aplicação e Análise de Indicadores Econômicos para o Brasil em 2005 (Using Data from the System of National Accounts to Estimate Input-Output Matrices: An Application Using Brazilian Data for 2005)". In: *Available at SSRN 1836495*.

Hajat, Anjum, Hsia, Charlene, and O'Neill, Marie S (2015). "Socioeconomic disparities and air pollution exposure: a global review". In: *Current environmental health reports* 2.4, pp. 440–450.

Hassett, Kevin A, Mathur, Aparna, and Metcalf, Gilbert E (2007). *The incidence of a US carbon tax: A lifetime and regional analysis*. Tech. rep. National Bureau of Economic Research.

Hoffmann, Rodolfo (2010). "Desigualdade da renda e das despesas per capita no Brasil, em 2002-2003 e 2008-2009, e avaliação do grau de progressividade ou regressividade de parcelas da renda familiar". In: *Economia e sociedade* 19.3, pp. 647–661.

IMF (2019). *Fiscal Policies for Paris Climate Strategies-from Principle to Practice.* Policy Paper. Fiscal Affairs Department and IMF e-Library - York University. International Monetary Fund. url: https://books.google.it/books? id= ezFKzQEACAAJ.

IPCC, Intergovernmental Panel on Climate Change (2007). "The physical science basis. Contribution of working group I to the fourth assessment report of the IPCC". In: *Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA* 996, p. 2007.

Kay, John A (1980). "The deadweight loss from a tax system". In: *Journal of Public Economics* 13.1, pp. 111–119.

Kerkhof, Annemarie C et al. (2008). "Taxation of multiple greenhouse gases and the effects on income distribution: A case study of the Netherlands". In: *Ecological Economics* 67.2, pp. 318–326.

Klenert, David et al. (2018). "Making carbon pricing work for citizens". In: *Nature Climate Change* 8.8, pp. 669–677.

Labandeira, Xavier, Labeaga, José M, and López-Otero, Xiral (2017). "A meta- analysis on the price elasticity of energy demand". In: *Energy Policy* 102, pp. 549– 568.

Labandeira, Xavier, Labeaga, José M, and Rodríguez, Miguel (2006). "A residential energy demand system for Spain". In: *The Energy Journal*, pp. 87–111.

Lazaridis, Panagiotis (2003). "Household meat demand in Greece: A demand systems approach using microdata". In: *Agribusiness* 19.1, pp. 43–59.

Lecocq, Sébastien and Robin, Jean-Marc (2015). "Estimating almost-ideal demand systems with endogenous regressors". In: *The Stata Journal* 15.2, pp. 554–573.

Maciel, Leticia dos Santos Benso et al. (2020). "Evaluating Public Policies for Fair Social Tariffs of Electricity in Brazil by Using an Economic Market Model". In: *Energies* 13.18, p. 4811.

Magalhaes, Aline Souza, Domingues, Edson Paulo, et al. (2013). *Economia de baixo carbono no Brasil: alternativas de políticas e custos de redução de emissões de gases de efeito estufa.* Tech. rep. Texto para discussão 491.

Mathur, Aparna and Morris, Adele C (2014). "Distributional effects of a carbon tax in broader US fiscal reform". In: *Energy Policy* 66, pp. 326–334.

Metcalf, Gillbert E and Weisbach, David (2009). "The design of a carbon tax". In: *Harv. Envtl. L. Rev.* 33, p. 499.

Mohring, Herbert (1971). "Alternative welfare gain and loss measures". In: *Economic Inquiry* 9.4, pp. 349–368.

Montoya, Marco Antonio, Lopes, Ricardo Luis, and Guilhoto, Joaquim José Martins (2014). "Desagregação setorial do balanço energético nacional a partir dos dados da matriz insumo-produto: uma avaliação metodológica". In: *Economia aplicada* 18.3, pp. 379–419.

Nemet, Gregory F, Holloway, Tracey, and Meier, Paul (2010). "Implications of incorporating air-quality co-benefits into climate change policymaking". In: *Envi-*

ronmental Research Letters 5.1, p. 014007.

Nijdam, Durk S et al. (2005). “Environmental load from Dutch private consumption: how much damage takes place abroad?” In: *Journal of Industrial Ecology* 9.1-2, pp. 147–168.

Nikodinoska, Dragana and Schröder, Carsten (2016). “On the emissions–inequality and emissions–welfare trade-offs in energy taxation: Evidence on the German car fuels tax”. In: *Resource and Energy Economics* 44, pp. 206–233.

Nordhaus, William D (1993). “Optimal greenhouse-gas reductions and tax policy in the” DICE” model”. In: *The American Economic Review* 83.2, pp. 313–317.

Nuaimy-Barker, Ravenna (2015). *G20 subsidies to oil, gas and coal production: Brazil*. Tech. rep. Overseas Development Institute (ODI).

OECD (2018). *Effective Carbon Rates 2018*. doi: https://doi.org/https://doi.org/10.1787/9789264305304-en.

Ohlendorf, Nils et al. (2018). “Distributional Impacts of Climate Mitigation Policies- a Meta-Analysis”. In: *DIW Berlin Discussion Paper No. 1776*.

Parry, Ian, Veung, Chandara, and Heine, Dirk (2015). “How much carbon pricing is in countries’ own interests? The critical role of co-benefits”. In: *Climate Change Economics* 6.04.

Pearce, David (1991). “The role of carbon taxes in adjusting to global warming”. In: *The economic journal* 101.407, pp. 938–948.

Pereda, Paula Carvalho et al. (2019). “Neutral carbon tax and environmental targets in Brazil”. In: *Economic Systems Research* 31.1, pp. 70–91.

Perobelli, Fernando Salgueiro, Faria, Weslem Rodrigues, and Almeida Vale, Vinicius de (2015). “The increase in Brazilian household income and its impact on CO2 emissions: Evidence for 2003 and 2009 from input–output tables”. In: *Energy Economics* 52, pp. 228–239.

Pindyck, Robert S (2013). “Climate change policy: what do the models tell us?” In: *Journal of Economic Literature* 51.3, pp. 860–72.

Pizer, William A and Sexton, Steven (2017). *Distributional impacts of energy taxes*. Tech. rep. National Bureau of Economic Research.

Poi, Brian P et al. (2012). “Easy demand-system estimation with quaids”. In: *Stata Journal* 12.3, p. 433.

Pollak, Robert A and Wales, Terence J (1981). “Demographic variables in demand analysis”. In: *Econometrica: Journal of the Econometric Society*, pp. 1533–1551.

Ramstein, Celine et al. (2019). *State and Trends of Carbon Pricing 2019*. Washington, DC: The World Bank. url: https://openknowledge.worldbank.org/handle/10986/31755.

Rao, Shilpa et al. (2016). “A multi-model assessment of the co-benefits of climate mitigation for global air quality”. In: *Environmental Research Letters* 11.12, p. 124013.

Ravallion, Martin (1992). *Poverty comparisons: a guide to concepts and methods*. The World Bank.

Reaños, Miguel A Tovar and Wölfing, Nikolas M (2018). “Household energy prices and inequality: Evidence from German microdata based on the EASI demand system”. In: *Energy Economics* 70, pp. 84–97.

Renner, Sebastian (2018). “Poverty and distributional effects of a carbon tax in Mexico”. In: *Energy Policy* 112, pp. 98–110.

Sajeewani, Disna, Siriwardana, Mahinda, and Mcneill, Judith (2015). “Household distributional and revenue recycling effects of the carbon price in Australia”. In: *Climate Change Economics* 6.03, p. 1550012.

Santos, Luan et al. (2018). “Impacts of carbon pricing on Brazilian industry: Domestic vulnerability and international trade exposure”. In: *Sustainability* 10.7, p. 2390.

Shonkwiler, J Scott and Yen, Steven T (1999). “Two-step estimation of a censored system of equations”. In: *American Journal of Agricultural Economics* 81.4, pp. 972–982.

Silva Freitas, Lucio Flavio da et al. (2016). “The distributional effects of emissions taxation in Brazil and their implications for climate policy”. In: *Energy Economics* 59, pp. 37–44.

Silveira Bezerra, Paula Borges da et al. (2017). “The power of light: socio-economic and environmental implications of a rural electrification program in Brazil”. In: *Environmental Research Letters* 12.9, p. 095004.

Stiglitz, Joseph E et al. (2017). *Report of the high-level commission on carbon prices*. Tech. rep.

Tiezzi, Silvia (2005). “The welfare effects and the distributive impact of carbon taxation on Italian households”. In: *Energy Policy* 33.12, pp. 1597–1612.

Timilsina, Govinda R (2018). *Where is the carbon tax after thirty years of research?* Tech. rep. Policy Research Working Paper. The World Bank.

Tomás, Manuel, López, Luis Antonio, and Monsalve, Fabio (2020). “Carbon footprint, municipality size and rurality in Spain: Inequality and carbon taxation”. In: *Journal of Cleaner Production*, p. 121798.

Wang, Qian et al. (2016). “Distributional effects of carbon taxation”. In: *Applied energy* 184, pp. 1123–1131.

Wang, Qian et al. (2019). “Distributional impact of carbon pricing in Chinese provinces”. In: *Energy Economics* 81, pp. 327–340.

West, J Jason et al. (2013). “Co-benefits of mitigating global greenhouse gas emissions for future air quality and human health”. In: *Nature climate change* 3.10, pp. 885–889.

Wier, Mette et al. (2005). “Are CO2 taxes regressive? Evidence from the Danish experience”. In: *Ecological economics* 52.2, pp. 239–251.

World Bank, WB (2020). *State and Trends of Carbon Pricing 2020*. Tech. rep. Washington, DC.

Yen, Steven T, Kan, Kamhon, and Su, Shew-Jiuan (2002). “Household demand for fats and oils: two-step estimation of a censored demand system”. In: *Applied Economics* 34.14, pp. 1799–1806.

6 Appendix A - Tables and Figures

Table A1: CO_2e emission coefficients (tCO_2e/USD million, 2019) from intermediate consumption and final demand for each good/service in the economy (2010)

Group	Good/service (IO)	Int. Consump.	Final Demand
4	Cargo transportation	906	1,096
4	Passenger transportation	880	1,065
4	Air transportation	482	520
4	Water transportation	422	519
8	Wood products, exclusive furniture	263	418
8	Paper, paperboard and paper articles	262	418
7	Printing services	245	392
8	Cement, plaster	155	282
8	Glass, ceramics and others	148	273
8	Metal products, excl. machines and equipment	95	258
1	Pork	11	171
1	Processed fish	11	171
2	Tabacco products	11	171
7	Animal feed	11	170
1	Other dairy Products	13	170
1	Coffee (processed)	11	170
1	Beef and other meat products	11	170
1	Beverages	11	170
1	Canned fruits, vegetables and fruit juices	11	170
8	Non-metallic Minerals	83	168
6	Coal	80	164
1	Sugar	64	163
1	Products derived from wheat, manioc and maize	17	163
5	Perfumery, soaps and cleaning products	42	163
1	Sterilized and pasteurized milk	10	162
1	Processed rice and products	16	162
7	Advertising and other technical services	101	161
1	Poultry	10	160
1	Oils and fats, vegetable and animal	10	160
5	Other chemical products	38	159
8	Inorganic chemicals	39	156
8	Paints, varnishes, enamels and lacquers	37	156

Note: This table presents the $CO2_2e$ coefficient estimates, in 2019 values, based on data from SUTs (2010) and the Brazilian Energy Mix (2010). Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil (3.9457 R$/USD).

Table A1: CO_2e emission coefficients (tCO_2e/USD million, 2019) from intermediate consumption and final demand for each good/service in the economy (2010) (cont.)

Group	Good/service (IO)	Int. Consump.	Final Demand
8	Plastic articles	36	154
8	Pesticides and disinfectants	34	152
8	Rubber articles	33	151
1	Other food products	10	151
8	Forestry	85	147
5	Pharmaceutical products	31	145
1	Sugarcane	82	144
1	Fisheries	82	144
1	Soybeans	82	144
1	Poultry and eggs	82	144
1	Citrus fruits	82	144
1	Swine by-products	82	144
1	Milk (cows and other animals)	82	144
1	Bovine animals	82	144
1	Other (permanent) agriculture	82	144
1	Other (temporary) agriculture	82	144
1	Corn	81	143
1	Rice	80	141
6	Ethanol and other biofuels	25	138
8	Goods of other enterprises	15	118
9	Other machines and mechanical equipment	13	117
6	Electricity, gas and other utilities	34	112
6	Gasoalcool	34	112
6	Diesel - biodiesel	32	112
8	Other Refined Petroleum Products	33	110
4	Aircraft, boats and other transport equipment	8	110
7	Electrical machinery and equipment	7	109
4	Trucks and buses	7	109
9	Equip. for measurement, testing	7	109
7	Home appliances	8	109
7	Furniture	8	108
7	Electronic material and communication equip.	7	108

Note: This table presents the $CO22e$ coefficient estimates, in 2019 values, built based on data from SUTs (2010) and the Brazilian Energy Mix (2010). Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil (3.9457 R$/USD).

Table A1: CO_2e emission coefficients (tCO_2e/USD million, 2019) from intermediate consumption and final demand for each good/service in the economy (2010) (cont.)

Group	Good/service (IO)	Int. Consump.	Final Demand
4	Automobiles, trucks and commercial vehicles	7	108
9	Office machines and equipment	7	108
3	Yarn and textile fibers	15	94
3	Textiles for household use and others	10	93
3	Textiles and leather products	9	91
3	Footwear and leather goods	9	90
3	Clothing articles and accessories	8	89
7	Rent and real estate services	21	80
4	Warehousing and others	23	72
7	Wholesale trade and sale, except motor vehicles	2	51
7	Maintenance of computers, telephones	1	49
9	Other administrative services	2	49
7	Intellectual Property Assets	1	49
4	Trade and repair of vehicles	1	49
2	Accommodation services, hotels and similar	2	48
7	Courier and other delivery services	1	48
7	Employer organizations, trade and union	1	48
7	Condominium and building management	1	48
7	Surveillance, security and investigation services	1	48
7	Domestic services	1	48
7	Personal services	1	48
7	Telecommunications and others	1	48
2	Film, music, radio	1	48
7	Financial services, insurance	1	48
2	Food services	1	48
9	Systems dev. and IT services	1	48
2	Books, newspapers and magazines	1	48
2	Arts, culture and recreation services	1	48
7	Architectural and engineering services	1	48
5	Private health	1	48
7	Legal services, accounting	1	47
2	Private education	1	47
9	Public administration	3	26
7	Water, sewage	3	26

Note: This Table presents the CO22e coefficient estimates, in 2019 values, based on data from SUTs (2010) and the Brazilian Energy Mix (2010). Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil (3.9457 R$/USD).

Table A2: Description of expenditure groups

Groups	Items
1 Food and beverages	Food, beverages and catering
2 Recreation, culture and education	Private education, arts, books, hotels
3 Clothing and footwear	Clothes, shoes, fabrics, textiles
4 Commuting and transportation	Air, water and ground transportation
5 Health and hygiene	Pharmaceutical products, private health
6 Energy	Electricity and Gas, Gasoline, Ethanol, Diesel and Charcoal
7 Housing	Residential appliances, rent, water and sewage
8 Other goods	Plastic, ceramic, wood and paper articles
9 Other services	Public and other administrative services

Note: This table presents the 9 consumption groups used in the demand system estimation, and it covers the total household consumption basket. Residential appliances include equipment such as stoves, washing machines, refrigerators, televisions, vacuum cleaners, electric ovens, electric irons, TVs, air conditioners, fans, computers, microwave ovens and clothes dryer. We are not able to split electricity from gas since these products are aggregated on the SUTs.

Table A3: Positive consumption and budget shares by group/ income level (%) (mean/(sd))

Groups	% of positive consumption			Budget share		
	All sample	20% richer	20% poorer	All sample	20% richer	20% poorer
Food/Bev.	89%	80%	94%	25.8%	15.7%	37.0%
				(23.0%)	(15.1%)	(26.7%)
Rec./Educ.	20%	40%	11%	4.1%	10.5%	1.0%
				(2.4%)	(8.5%)	(0.9%)
Cloth./Foot.	77%	79%	63%	3.8%	3.2%	3.1%
				(3.7%)	(1.8%)	(2.4%)
Com./Transp.	37%	38%	21%	7.3%	12.9%	3.7%
				(6.4%)	(12.1%)	(3.4%)
Health/Hyg.	18%	46%	17%	9.5%	7.6%	11.8%
				(9.4%)	(5.9%)	(10.0%)
Energy	65%	78%	41%	14.4%	21.7%	9.3%
				(9.1%)	(19.5%)	(6.8%)
Housing	85%	91%	79%	14.8%	16.3%	12.8%
				(14.2%)	(13.0%)	(5.8%)
Oth.goods	76%	47%	84%	12.3%	6.0%	18.0%
				(11.7%)	(3.0%)	(15.0%)
Oth.services	30%	42%	19%	3.2%	6.1%	1.3%
				(2.9%)	(5.9%)	(0.6%)

Note: This table shows the descriptive statistics of budget shares (in 2019 USD), positive consumption for each of the nine groups and income levels. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Other goods contain rubber, plastic, ceramic goods, non-metallic minerals, inorganic chemicals. Other services include development of systems and other information services, private and public administrative services.

Table A4: Summary statistics: socio-economic characteristics (Mean/(sd))

Variables	All sample	20% richest	20% poorest
Education - household head (years)	7.4	10.6	5.9
	(0.03)	(0.07)	(0.03)
Age - household head (years)	47.6	49.1	42.6
	(0.15)	(0.24)	(0.21)
Female headed households (%)	31.5	25.9	36.9
	(46.4)	(43.8)	(48.2)
Bathrooms (Number)	1.3	2.8	1.1
	(0.01)	(0.02)	(0.01)
Rooms (Number)	3.3	7.5	3.1
	(0.00)	(0.02)	(0.00)
People in the household (Number)	3.4	3.5	3.1
	(1.73)	(1.50)	(1.80)
Home ownership (%)	69.5	73.0	65.7
	(45.9)	(44.3)	(47.4)
Total earnings (USD 2019/per capita/year)	15,304	46,492	3,335
	(14,275)	(41,419)	(1,117)
Disposable income (USD 2019/per capita/year)	6,185	15,887	1,468
	(3,909)	(9,702)	(553)
Car ownership (%)	27.7	76.3	5.3
	(44.7)	(42.5)	(22.3)
Moto ownership (%)	16.2	20.9	10.0
	(36.8)	(40.6)	(30.0)
Electricity (%)	97.0	99.5	93.4
	(17.0)	(6.9)	(24.8)
Residential appliances (Number)	8.0	13.0	5.0
	(4.0)	(6.0)	(2.0)

Note: This table presents the descriptive statistics for control variables according to income-level of socioeconomic variables used in the demand system. Residential appliances include stoves, freezers, refrigerators, vacuum cleaners, electric ovens, electric irons, washing machines, color TVs, black and white TVs, sound systems, radios, air conditioners, fans, computers, microwaves, DVD player, clothes dryers and washing machines. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil.

Table A5: Price Indices by group and income level (in USD 2019 values) (Mean/(sd))

Groups	All sample	20% richer	20% poorer
Food/Bev. (USD/Kg)	9.48	11.22	8.39
	(0.06)	(0.23)	(0.06)
Rec./Educ (USD/service)	250.46	428.40	163.14
	(1.99)	(6.80)	(1.54)
Cloth./Foot (USD/item)	31.39	38.27	27.47
	(0.12)	(0.39)	(0.15)
Com./Transp. (USD/service)	512.05	694.08	390.82
	(3.58)	(11.13)	(3.61)
Health/Hyg. (USD/service)	9.94	14.22	7.48
	(0.09)	(0.37)	(0.04)
Energy (USD/KWh,L)	15.93	20.55	12.89
	(0.09)	(0.31)	(0.11)
Housing (USD/service)	172.41	242.35	134.76
	(1.32)	(4.57)	(1.22)
Oth. goods (USD/item)	55.10	83.96	38.23
	(0.63)	(2.03)	(0.53)
Oth. services (USD/service)	117.10	191.42	72.60
	(0.92)	(2.94)	(0.98)

Note: This table presents price indices by group and income level, converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil. They should be interpreted as relative price indexes. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Other goods contain rubber, plastic, ceramic goods, non-metallic minerals, inorganic chemicals. Other services include development of systems and other information services, private and public administrative services.

Table A6: Total CO_2e emission (tCO_2e/hh/year, in 2019 values)

Groups	Average	20% richest	20% poorest
Food/Beverages	3.3	6.0	0.4
Recreation/education	1.6	3.1	0.0
Clothing/footwear	0.3	0.8	0.1
Commuting/Transportation	2.0	8.0	0.1
Health/Hygiene	0.6	1.8	0.2
Energy	1.2	3.0	0.3
Housing	0.6	1.3	0.3
Other goods	0.6	2.0	0.3
Other services	0.2	0.1	0.0
Total	10.5	26.1	1.7

Note: This table presents the $CO2_2e$ coefficient estimates, in 2019 values, based on data from SUTs (2010) and the Brazilian Energy Mix (2010). Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil.

Table A7: Expenditure and price elasticities estimated using censored QUAIDS model (mean/(sd)): full sample, 20% poorest, and 20% richest.

Groups	Full Sample			20% richest			20% poorest		
	η_i	E_{ii}	E^{H}_{ii}	η_i	E_{ii}	E^{H}_{ii}	η_i	E_{ii}	E^{H}_{ii}
1 Food/bev.	0.841***	-0.875***	-0.658***	0.763***	-1.051*	-0.914***	0.490***	-0.829***	-0.463***
	(0.000)	(0.029)	(0.058)	(0.110)	(0.082)	(0.083)	(0.110)	(0.143)	(0.132)
2 Recr./educ.	1.641***	-1.237***	-1.171***	0.656***	-0.615**	-0.472***	3.927***	-1.438***	-1.429***
	(0.017)	(0.019)	(0.036)	(0.073)	(0.073)	(0.073)	(0.071)	(0.108)	(0.106)
3 Cloth/Foot.	2.502***	-1.337***	-1.242***	1.255***	-1.173**	-1.011***	2.036***	-1.500***	-1.461***
	(0.005)	(0.028)	(0.055)	(0.071)	(0.110)	(0.108)	(0.088)	(0.127)	(0.123)
4 Comm./Tranp.	0.836***	-0.576***	-0.581***	0.976***	-1.119**	-1.093***	1.981***	-0.421***	-0.385***
	(0.013)	(0.049)	(0.097)	(0.055)	(0.078)	(0.080)	(0.068)	(0.164)	(0.153)
5 Health/Hyg.	1.208***	-1.295***	-1.180***	0.304***	-0.826	-0.771***	1.954***	-1.611***	-1.494***
	(0.018)	(0.032)	(0.063)	(0.070)	(0.051)	(0.049)	(0.081)	(0.081)	(0.078)
6 Energy	1.826***	-0.931***	-0.761***	1.792***	-1.081*	-1.154***	1.942***	-0.833***	-0.747***
	(0.005)	(0.026)	(0.050)	(0.064)	(0.045)	(0.045)	(0.076)	(0.061)	(0.061)
7 Housing	0.509***	-0.480	0.315***	0.258***	-1.143**	-0.917***	0.572***	-0.808***	-0.657***
	(0.011)	(0.022)	(0.043)	(0.088)	(0.051)	(0.054)	(0.089)	(0.068)	(0.069)
8 Other goods	0.734***	-0.741***	-0.650***	0.252***	-1.081*	-0.998*	1.027***	-0.336***	-0.152***
	(0.012)	(0.020)	(0.039)	(0.093)	(0.075)	(0.071)	(0.134)	(0.174)	(0.169)
9 Other servic.	1.482***	-1.305*	-1.257	0.967***	-0.783**	-0.715**	3.855***	-1.352	-1.341
	(0.114)	(0.034)	(0.067)	(0.063)	(0.073)	(0.073)	(0.076)	(0.065)	(0.065)

Note: This table presents expenditure (η_i) and uncompensated (E_{ii}) and compensated (E^{H}_{ii}) price elasticities for group estimates, calculated at the sample mean using censored QUAIDS. Neighboring prices and disposable income are used as instruments to calculate final household price indexes and expenditures. Standard errors in parentheses. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil. $*$ p-value$<$0.10, $**$ p-value $<$ 0.05, $***$ p-value$<$0.01. Diagonals represent the own-price elasticities.

Table A8: Marshallian (E_{ij}) price elasticities using censored QUAIDS, full sample (Mean/ (sd))

Hicksian	1		2		3		4		5		6		7		8		9	
1 Food/bev.	-0.875	***	-0.466	***	-0.213	***	1.753	***	-0.444	***	-0.118	***	-0.066	**	0.221	***	-0.534	***
	(0.029)		(0.030)		(0.019)		(0.087)		(0.075)		(0.031)		(0.030)		(0.036)		(0.050)	
2 Recr./educ.	0.146	***	-1.237	***	-0.032	***	0.928	***	-0.295	***	-0.196	***	-0.073	***	0.291	***	-0.295	***
	(0.015)		(0.019)		(0.005)		(0.051)		(0.032)		(0.017)		(0.015)		(0.021)		(0.028)	
3 Cloth/foot	0.192	***	-0.250	***	-1.337	***	1.122	***	-0.017	***	-0.017		-0.188	***	0.683	***	-0.472	***
	(0.030)		(0.026)		(0.028)		(0.086)		(0.066)		(0.027)		(0.031)		(0.031)		(0.042)	
4 Comm./Transp.	0.187	***	-0.215	***	0.092	***	-0.576		0.010	***	-0.148	***	-0.143	***	0.208	***	-0.296	***
	(0.011)		(0.012)		(0.010)		(0.049)		(0.023)		(0.011)		(0.009)		(0.012)		(0.016)	
5 Health/Hygiene	0.183	***	-0.237	***	-0.114	***	1.233	***	-1.295	***	-0.175	***	-0.067	***	0.291	***	-0.313	***
	(0.018)		(0.019)		(0.011)		(0.069)		(0.032)		(0.020)		(0.017)		(0.021)		(0.032)	
6 Energy	0.132	***	-0.436	***	0.110	***	0.702	***	-0.532	***	-0.931	***	-0.395	***	0.115	***	-0.148	***
	(0.021)		(0.019)		(0.019)		(0.081)		(0.054)		(0.026)		(0.021)		(0.021)		(0.029)	
7 Housing	0.292	***	-0.211	***	0.030		0.243	***	0.029		-0.382	***	-0.480	***	-0.034		-0.100	**
	(0.016)		(0.015)		(0.020)		(0.062)		(0.041)		(0.015)		(0.022)		(0.015)		(0.021)	
8 Other goods	0.140	***	-0.244	***	0.088	***	1.054	***	-0.476	***	-0.194	***	-0.373	***	-0.741	***	-0.199	***
	(0.016)		(0.017)		(0.008)		(0.065)		(0.033)		(0.018)		(0.015)		(0.020)		(0.028)	
9 Other serv.	0.125	***	-0.307	***	-0.095	***	1.009	***	-0.345	***	-0.110	***	-0.048	***	0.289	**	-1.305	*
	(0.017)		(0.019)		(0.006)		(0.054)		(0.037)		(0.018)		(0.017)		(0.023)		(0.034)	

Note: This table presents the Marshallian (uncompensated) price elasticities estimates, calculated at the sample mean using censored QUAIDS. Neighboring prices and disposable income are used as instruments to calculate final household price indexes and expenditures. Standard errors in parentheses. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil. $*$ p-value<0.10, $**$ p-value < 0.05, $***$ p-value<0.01. Diagonals represent the own-price elasticities.

Table A9: Hicksian (E^H_{ij}) price elasticities using censored QUAIDS, full sample (Mean/ (sd))

Hicksian	1		2		3		4		5		6		7		8		9	
1 Food/bev.	-0.658	*	-0.043	***	0.431	***	1.734	***	-0.133	***	0.187	***	0.222	**	0.410	***	-0.152	**
	(0.022)		(0.003)		-(0.039)		(0.086)		(0.022)		(0.016)		(0.054)		(0.067)		(0.014)	
2 Recr./educ.	0.180	***	-1.171	***	0.069	***	0.925	***	-0.246	***	-0.148	***	-0.028	***	0.321	***	-0.235	**
	(0.019)		(0.018)		(0.005)		(0.051)		(0.027)		(0.013)		(0.006)		(0.023)		(0.022)	
3 Cloth/foot.	0.224	*	-0.188	***	-1.242	***	-1.242	***	-0.392	**	0.028		-0.145	***	0.711	***	-0.416	***
	(0.035)		(0.013)		(0.026)		-(0.095)		(1.519)		(0.422)		(0.016)		(0.032)		(0.037)	
4 Comm./Transp.	0.249	***	-0.095	***	0.274	***	-0.581		0.098		-0.062		-0.062	***	0.262	***	-0.188	***
	(0.014)		(0.005)		(0.029)		(0.050)		(0.236)		(0.303)		(0.004)		(0.016)		(0.010)	
5 Health/ Hygiene	0.263	***	-0.081		0.123	***	1.226	***	-1.180	***	-0.063	***	0.039	***	0.361	***	-0.172	***
	(0.026)		(0.115)		-(0.012)		(0.169)		(0.029)		(0.007)		-(0.010)		(0.026)		(0.018)	
6 Energy	0.253	***	-0.200	***	0.471	***	0.692	***	-0.358		-0.761	***	-0.234	***	0.220		0.065	***
	(0.040)		(0.009)		(0.081)		(0.080)		(0.037)		(0.021)		(0.012)		(0.243)		(0.027)	
7 Housing	0.416	***	0.032	***	0.400		0.232	***	0.208		-0.207	***	-0.315	***	0.074		0.119	**
	(0.022)		-(0.002)		(0.277)		(0.059)		(0.292)		(0.008)		(0.014)		(0.016)		(0.013)	
8 Other goods	0.244	***	-0.042	***	0.395	***	1.045	***	-0.327		-0.049	***	-0.235	***	-0.650	***	-0.016	***
	(0.028)		(0.003)		(0.036)		(0.064)		(0.023)		(0.004)		(0.010)		(0.017)		(0.002)	
9 Other serv.	0.152	***	-0.254	***	-0.014	***	1.007	***	-0.306		-0.071	***	-0.012	***	0.313	**	-1.257	***
	(0.020)		(0.015)		(0.001)		(0.054)		(0.032)		(0.012)		(0.004)		(0.025)		(0.033)	

Note: This table presents the Hicksian (compensated) price elasticities estimates, calculated at the sample mean using censored QUAIDS. Neighboring prices and disposable income are used as instruments to calculate final household price indexes and expenditures. Standard errors in parentheses. Income deciles are constructed based on total household monetary and non-monetary income reported by POF. Monetary values were converted to USD 2019 using the average exchange rate for that year from the Central Bank of Brazil. $*$ p-value<0.10, $**$ p-value < 0.05, $***$ p-value<0.01. Diagonals represent the own-price elasticities.

Table A10: Distributional welfare and emission effects of a USD 40/tCO$_2$e and USD 80/tCO$_2$e carbon tax in Brazil (USD 2019 values)

	USD 40/tCO2e			USD 80/tCO2e		
	Full Sample	20% richest	20% poorest	Full Sample	20% richest	20% poorest
Emissions (MtCO2e)	-12.53	-7.58	-0.33	-25.19	-14.63	-0.66
Emissions (%)	-2.08%	-2.53%	-1.65%	-4.18%	-4.88%	-3.28%
DWL (Mi USD/year)	-237.90	-205.36	-21.17	-244.25	-209.47	-21.68
DWL (Mi USD/year) % expenditure	-0.02%	-0.04%	-0.02%	-0.02%	-0.04%	0.02%
EV (Mi USD/year)	-854.46	-308.78	-83.87	-874.48	-105.48	-85.75
EV (Mi USD/year) % expenditure	-0.07%	-0.06%	-0.10%	-0.07%	-0.06%	-0.10%
Tax Revenue (Mi USD/year)	616.21	103.42	62.70	629.85	103.99	64.07
Lump sum transfer (revenue neutral) (%/year)	0.05%	0.02%	0.07%	0.05%	0.02%	0.07%

Note: This table presents the results of a simulation exercise considering an economy-wide carbon tax. DWL= deadweight loss; EV= equivalent variation.

Table A11: Distributional welfare and emission effects of a USD 40/tCO_2e and USD 80/tCO_2e carbon tax levied on commuting and transportation services (USD 2019 values)

	USD 40/tCO2e			USD 80/tCO2e		
	Full Sample	20% richest	20% poorest	Full Sample	20% richest	20% poorest
Emissions (MtCO2e)	-2.70	-2.82	-0.03	-5.33	-5.46	-0.05
Emissions (%)	-0.45%	-0.94%	-0.13%	-0.89%	-1.82%	-0.25%
DWL (Mi USD/year)	-22.82	-15.37	0.26	-23.37	-15.84	-0.27
DWL (Mi USD/year) % expenditure	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
EV (Mi USD/year)	-49.65	-78.70	-2.01	-97.58	-81.13	-2.05
EV (Mi USD/year) % expenditure	0.00%	0.01%	0.00%	0.01%	0.01%	0.00%
Tax Revenue (Mi USD/year)	72.48	63.34	1.75	74.21	65.29	1.78
Lump sum transfer (revenue neutral) (%/year)	0.01%	0.01%	0.00%	0.01%	0.01%	0.00%

Note: This table presents the results of a simulation exercise considering a carbon tax applied only on commuting and transportation. DWL= deadweight loss; EV= equivalent variation.

Table A12: Distributional welfare and emission effects of a USD 40/tCO$_2$e and USD 80/tCO$_2$e carbon tax levied on food and beverages (USD 2019 values)

	USD 40/tCO2e			USD 80/tCO2e		
	Full Sample	20% richest	20% poorest	Full Sample	20% richest	20% poorest
Emissions (MtCO2e)	-5.16	-1.93	-0.02	-10.24	-3.78	-0.05
Emissions (%)	-0.86%	-0.64%	-0.12%	-1.70%	-1.26%	-0.25%
DWL (Mi USD/year)	-82.48	-6.38	-0.85	-84.83	-6.56	-0.86
DWL (Mi USD/year) % expenditure	0.01%	0.00%	0.00%	-0.01%	0.00%	0.00%
EV (Mi USD/year)	-441.37	-76.69	-4.21	-453.99	-92.00	-4.23
EV (Mi USD/year) % expenditure	0.03%	0.01%	0.00%	-0.04%	0.02%	0.00%
Tax Revenue (Mi USD/year)	358.89	70.31	3.35	369.15	85.44	3.37
Lump sum transfer (revenue neutral) (%/year)	0.03%	0.01%	0.00%	0.03%	0.02%	0.00%

Note: This Table presents the results of a simulation exercise considering a carbon tax applied only on food and beverages. DWL= deadweight loss; EV= equivalent variation.

Table A13: Distributional welfare and emission effects of a USD 40/tCO_2e and USD 80/tCO_2e carbon tax transferring the revenues from richest households to the poorest households in the form of social tariff for energy goods consumption (USD 2019 values)

	USD 40/tCO2e			USD 80/tCO2e		
	Full Sample	20% richest	20% poorest	Full Sample	20% richest	20% poorest
Emissions (MtCO2e)	-4.89	-7.58	-0.07	-9.96	-14.63	-0.10
Emissions (%)	-0.81%	-2.53%	-0.36%	-1.65%	-4.88%	-0.49%
DWL (Mi USD/year)	-237.90	-205.36	-21.17	-244.25	-209.47	-21.68
DWL (Mi USD/year) % expenditure	-0.02%	-0.04%	-0.02%	-0.02%	-0.04%	-0.02%
EV (Mi USD/year)	-812.86	-308.78	-19.61	-842.74	-105.48	-18.29
EV (Mi USD/year) % expenditure	-0.06%	-0.06%	-0.02%	-0.06%	-0.06%	-0.02%
Tax Revenue (Mi USD/year)	615.92	0.00	40.78	629.56	0.00	39.98
Lump sum transfer (revenue neutral) (%/year)	0.05%	0.00%	0.05%	0.05%	0.00%	0.05%

Note: This Table presents the results of a simulation exercise considering a carbon tax applied only on food and beverages. DWL= deadweight loss; EV= equivalent variation.

Table A14: Expenditure share of energy goods by income level

Mean / (sd)	All sample	20% richer	20% poorer
Charcoal	0.1%	0.3%	0.2%
	(0.1%)	(0.9%)	(0.2%)
Diesel	2.8%	4.1%	1.1%
	(2.1%)	(0.0%)	(0.0%)
Electricity	61.9%	40.5%	73.4 %
	(20.2%)	(20.5%)	(11.2%)
Ethanol	3.7%	5.2%	1.7%
	(10.4%)	(3.2%)	(0.9%)
Gasoline	32.7%	50.8 %	23.2%
	(30.7%)	(18.1%)	(9.6%)

Source: Prepared by the authors. Based on Household Budget Survey (2008/09).