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The distributional aspects of environmental quality and environmental policies

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1. Introduction

The growing public discontent with the distributional implications of policy reforms has brought inclusiveness concerns in the spotlight of policy-makers across a range of public policy domains. These concerns are supported by recent evidence on rising inequalities in many OECD countries (Piketty 2013; OECD 2018a; OECD 2018b). Inequality, in many ways, may limit economic growth, for example by harming human health and education (e.g. Ostry et al. 2014), undermine public trust and ultimately compromise the ability of governments to ensure an efficient provision of public goods (including environmental quality). Despite the urgency to address major global environmental challenges (OECD 2017a) and to re-double actions towards achieving a greener growth (OECD 2017b), such efforts have been increasingly compromised in many countries, in part due to perceptions that they "unfairly" impact certain groups of individuals.

Consequently, social inclusivity is increasingly a related objective of environmental policies. Policies perceived as "fair" are more likely to be accepted, and complied with, by the public at large (OECD 2017c). This helps introduce more ambitious environmental policies and accelerate their enforcement. The latter is important because stronger incentives are needed to induce further innovation (OECD 2011a, 2012a) and to transition faster to greener growth (see e.g. OECD 2015 for a discussion of the economic benefits of moving early on climate policy). A complementary motivation for promoting policy inclusiveness is that mitigating potentially regressive impacts of environmental policies can, in turn, contribute to alleviating other aspects of inequality such as health, education or economic development.

Environmental policy is a domain where resolving these tensions is challenging because environmental externalities are difficult to value in market terms and these non-market impacts (e.g. on climate, biodiversity, ecosystem services) are often diffuse, occur far away or over large areas, and over long time periods. Consequently, environmental policy faces the difficulty of identifying its non-market benefits (enjoyed by certain victims of pollution), while its market costs (borne by certain polluters) are clearly 'visible'. Improvements in environmental quality are thus often disregarded as a source of inequality because of the inherent difficulty of reflecting environmental externalities in market values.

Moreover, given that environmental quality varies across space and over time in important ways, the benefits of environmental policies are unlikely to be evenly distributed across individuals within a given population. Policy makers increasingly seek to address potential distributional impacts of environmental policies, however, such efforts must not undermine the underlying environmental incentives, as shall be discussed below.

Environmental quality varies by location. One relevant question is whether this variation is correlated with socio-economic characteristics of populations (measured via income, education etc.), and particularly whether poorer groups may be disproportionately exposed or impacted by poor environmental quality. Systematic efforts to monitor this are relatively scarce among OECD governments with approximately half of OECD countries indicating that such efforts are currently underway.[1] This note reviews the evidence on the socioeconomic distribution of exposure to a selection of common environmental risks such as air and noise pollution and natural hazards, and of access to amenities such as green

[1] Based on a survey of the OECD Working Party on Environmental Information.

space. It finds that there are many examples where disadvantaged groups within countries, regions or cities are exposed to greater environmental risks, and ultimately greater potential harm or damages, or have poorer access to amenities than more advantaged groups. However, this pattern is not ubiquitous, there are examples where no difference, or the opposite difference is observed.

Another set of questions concerns environmental policy[2] and its potential to mitigate or exacerbate the distribution of environmental quality, and inequality more generally. The challenge here is not to achieve a perfectly uniform distribution of impacts; as alluded above, environmental policies will necessarily lead to heterogeneous impacts. Rather, the objective is to prevent that certain segments of society, such as the poor or the socially fragile, are subject to a *disproportionally* large burden (see e.g., Pearce 2006). This note reviews the recent empirical literature and finds that although the evidence on the direct effects of environmental policy varies by policy instrument type, the general lesson is that well-designed environmental policy reforms incorporating appropriately targeted compensation schemes (e.g. within existing social welfare systems) can neutralise any direct negative distributional outcomes and generate progressive outcomes overall. Depending on the design, poor households can even enjoy net gains from such policy reforms.

The remainder of this note is structured in two parts. First, the empirical evidence on the distribution of environmental quality is reviewed (Section 2). Next, the empirical evidence on the distributional effects of environmental policies is reviewed (Section 3). In both cases, the reviews presented here are not meant to be exhaustive but rather aim at illustrating some of the key findings in the literature. The paper concludes by highlighting good practices and remaining open questions for further research (Section 4).

[2] E.g. carbon taxes, municipal solid waste management charges, renewable energy tax credits, energy efficiency investment subsidies, motor vehicle emission standards, etc.

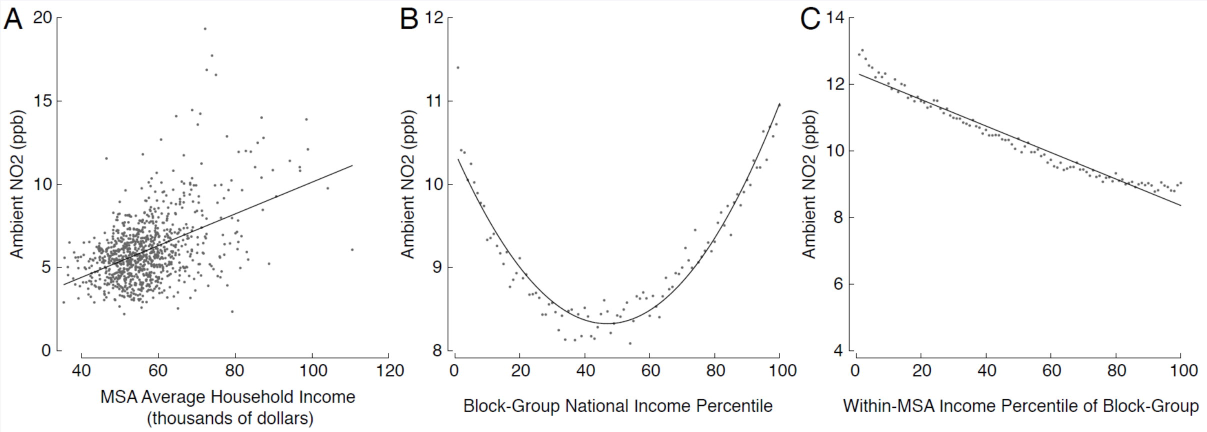
2. Evidence on the distribution of environmental quality

Exposure to environmental externalities varies between and within countries. Since the early 1970s researchers and policy makers have questioned whether some populations disproportionately bear the burden of environmental damages, with a particular concern for the poor populations. Empirical evidence reviewed here broadly supports this concern however the difference is sometimes very small and exceptions (where no difference, or the opposite is observed) abound.

The socioeconomic distribution of environmental quality varies by the spatial scale and the unit of analysis used

Relationships between socioeconomic status (e.g., based on income, education, etc.) and environmental quality can be observed at different spatial scales (countries, regions, city neighbourhoods) and also across different societal groups within a population. When reading this section it is important to recognise that conclusions made using a particular unit of analysis may not hold for another. Box 2.1 illustrates the importance of scale and choice of geographical unit of analysis by showing how the relationship between household income and exposure to nitrogen dioxide (NO_2) in the United States varies when using metropolitan areas, census blocks and gridded NO_2 concentration data. This complexity is one of the reasons why it is difficult to draw general conclusions from the body of empirical evidence.

Box 2.1. Exposure to NO_2 by household income at different units of analysis in the USA



Source: Hsiang et al. (2018)

As an illustrative example, Hsiang et al. (2018) show that at the national level across the 932 Metropolitan Statistical Areas (MSAs) nitrogen dioxide (NO_2) concentrations increase with average household income (Panel A). This is because NO_2 concentrations increase with population density and households in higher-density large cities are generally higher-earning.

However, when the unit of analysis is changed to all census block groups (small census aggregation units that cover the entire USA) binned by income percentile using national income (Panel B) a completely different U-shaped relationship emerges with the lowest and highest income census blocks tending to have higher concentrations, reflecting the fact that the more polluted larger cities are home to both the richest *and* poorest in society.

Lastly, when all census block groups within MSAs are binned by their income percentile in their respective MSA (Panel C) and these percentile bins are plotted against their population-weighted average NO_2 concentration, a third completely different relationship emerges and we can see that *within* MSAs, NO_2 concentrations decrease as census block average household incomes increase. The poorer areas of cities are more exposed than the richer areas.

This relatively simplistic example where radically different effects can be observed even at similar scales illustrates the importance of the effects of scale and of the choice of unit of analysis. The three conclusions are individually valid but none tell the whole story.

Between countries, exposure to environmental risk is strongly correlated with sociodemographic development (determined by using income, education and fertility rate) (GBD 2017). The greatest environmental risk in more developed countries comes from exposure to ambient outdoor air pollution whereas the greatest risks in less developed countries are indoor air pollution from solid fuel use and unsafe water supply. The harm caused by environmental risks is proportionally much greater in less developed countries, for example, it is estimated that exposure to all types of air pollution causes around 10% of premature deaths in less developed countries compared to 3-5% in more developed countries (GBD 2017).

Within countries, evidence of environmental inequalities between regions, between cities, between urban and rural areas and between neighbourhoods is mixed. Table 2.1 provides some examples of studies that examine the socioeconomic distribution of exposure to a non-exhaustive range of environmental risks and amenities. Overall, the evidence suggests that less advantaged groups frequently live in environments of poorer quality however this finding is not universal and sometimes the observed differences are small. Conclusions are possible only on a case-by-case basis for specific risks and specific countries, regions or cities following empirical study. For example, studies generally show that disadvantaged groups in the United States are exposed to modestly higher concentrations of fine particles ($PM_{2.5}$) than more affluent groups however this pattern is not found in all cities, nor for other pollutants. Where higher-income groups are found to be more exposed it is often where they are located in high-density city centres (Hajat et al. 2015). Results seem less nuanced for some dimensions of environmental quality: lower-income households self-report more noise nuisance in surveys in Germany and Switzerland and a report by Science for Environment Policy (2016) details a number of empirical studies that support this elsewhere. Furthermore, research shows that more advantaged groups in urban areas in developed countries virtually always have access to better quality parks than less advantaged groups (Rigolon 2016).

Table 2.1. Literature on the socioeconomic variation in exposure and vulnerability to different environmental risks

Studies have been grouped by type of exposure for convenience however several studies cover two or more kinds of risk (e.g. air and noise pollution).

Study & scope	Summary of results
Air quality	
Hajat et al. (2015) Review of 37 studies on socioeconomic status (SES) and air pollution at multiple scales. Primarily in Europe and North America with a small number of studies elsewhere, including New Zealand and Hong Kong. SES is typically determined using one or more metrics such as income, education, occupation, housing, access to mains water, electricity, possession of certain assets etc.	**North America:** Studies generally show low SES group are exposed to higher concentrations of pollutants. Exceptions exist in some large cities where high-SES groups cluster in polluted but otherwise desirable city centres. The magnitude of the difference varies but is generally quite modest.[3] **Europe:** Studies show mixed results that are nuanced by pollutant type, location, city size, and SES metric used. Several studies show similar patterns as observed in North America (lower-SES groups more exposed with exceptions in cities). Others find non-linear results where the richest and poorest were most exposed and the middle-income groups least exposed. **Rest of the world:** The limited studies covering the rest of the world generally show that deprived groups are exposed to greater pollution. Overall evidence shows that lower SES linked to higher exposure in much of the world. However, results are place-specific and exceptions abound.
Richardson et al. (2013) EU-wide study on regional differences in household income, PM_{10} exposure and health outcomes (NUTS 2 scale).	Poorer regions have higher exposure to PM10 at the European scale, driven by the West-East divide but this does not hold when Western and Eastern Europe are considered individually. Some evidence that low income populations are more sensitive to health effects.
Fecht et al. (2015) Association between SES, age, ethnicity and air pollution in England and the Netherlands. Multiple scales (national, regional and city-levels).	More deprived and more ethnically diverse neighbourhoods have higher PM10 and NO2 concentrations. This is driven by inequality in urban areas and the urban-rural contrast; little inequality was found in rural areas. Even though England and the Netherlands are somewhat similar, the magnitude of inequality varies considerably by country and region. Absolute differences in exposure are sometimes large (e.g. most deprived neighbourhoods in England are

[3] Health responses can be non-linear, meaning small differences in the 'dose' or pollution concentration can potentially have relatively large health consequences. See e.g. Apte et al., (2015) for an analysis of the PM2.5 concentration-response.

	exposed on average to 2.6µg/m^3 higher PM10 concentrations than the least deprived).
Milojevic et al. (2017) Association between SES, urban-rural differences, air pollution and mortality in England.	Concentrations of most pollutants modestly higher in deprived areas. SES differences in mortality are marginally attributable to differences in air quality (other factors are more important). Measures that cause a general reduction of background pollutant concentrations may help (very modestly) to narrow socioeconomic differences in health.
Jans et al. (2018) SES and children's health outcomes during high PM_{10} concentration episodes in Sweden.	Evidence that poorer children suffer greater respiratory health problems during episodes of high pollution, likely explained by differences in baseline health.
Pinault et al. (2016) SES and exposure to nitrogen dioxide in three large Canadian cities.	Deprivation is associated with higher exposure to NO_2 for some (but not all) SES variables notably non-native speakers and relationship status.
Noise	
Science for Environment Policy (2016) In-depth report on links between noise and air pollution and socioeconomic status (drawing on many of the same sources as this table).	Overall, evidence strongly suggests that more advantaged communities are less likely to suffer air and noise pollution related health impacts as poorer communities, even where the advantaged communities live in more polluted areas.
Swiss Health Observatory (Obsan) (2018) National Health of the Population survey.	Lower-income and lower-education groups report higher levels of noise nuisance. Overall noise nuisance has been declining since 2002. Similar patterns are observed for reported nuisance from air pollution from vehicle exhaust.
Federal Ministry of Environment and German Environment Agency (2016) National survey of Environmental awareness.	Environmental stress is more commonly reported in disadvantaged areas. Lower-income groups self-report as being considerably more affected by air and noise pollution that higher income groups.
Water quality and sanitation	
GBD (2016) Global systematic survey of the burden of disease including the contribution of different risks to mortality.	Populations of lower socio-demographic development (based on income, education and total fertility rate) are exposed to much higher risks from unsafe water (and indoor air pollution and to a lesser extent outdoor air pollution) and these risks account for a much greater share of overall

	mortality and morbidity compared to more developed countries.
Green space and contact with nature	
Mitchell and Popham (2008) Effect of exposure to natural environment on health inequalities in England.	Health inequalities related to deprivation in all-cause mortality and mortality from circulatory diseases lower in populations living in the greenest areas.
Rigolon (2016) Review of 49 studies of SES inequality in access to parks, mostly in developed countries.	'Striking' inequality in park *size* and *quality*. Low SES and ethnic minority groups have access to fewer acres of parks, fewer acres of parks per person, and to parks with lower quality, maintenance, and safety than more privileged groups.
Disamenities	
Martuzzi et al. (2010) Review of 47 studies examining the relationship between SES and proximity to waste management sites in Europe and the USA.	Waste management facilities more often disproportionately located in more deprived areas. The health consequences are difficult to estimate.
Natural hazards and climate	
Park et al. (2018) Distributional consequences of climate change with regard to exposure to temperature extremes using household surveys and weather data from 52 countries in Africa, Central and South America, Asia.	Poorer households are more exposed to extreme heat episodes than richer households in three-quarters of the countries studied. This relationship was most frequently observed in African countries and in countries that are generally hot. This is likely related to geography and not structural transformation (e.g. level of urbanisation or local land use). In cool countries the inverse is observed (poorer households live in cooler areas).
Hsiang et al. (2017) Costing of climate change scenarios in the USA. A simulation study	Losses (relative to per capita income) compared to a counterfactual no-change scenario are on average greater in counties that are already poor – under this model climate change increases pre-existing inequalities.
IPCC (2014) Report on Impacts, Adaptation, and Vulnerability to climate change. Synthesis report drawing on empirical findings.	People who are socially, economically, culturally, politically, institutionally, or otherwise marginalized are especially vulnerable to the climate-change related hazards like e.g. heat stress, extreme precipitation, flooding, landslides, air pollution, drought etc.

Evidence of the benefits or harm (typically to health) associated with certain amenities or risks such as exposure to air pollution is well established (see e.g., Apte et al. 2015 for the cardiovascular and respiratory disease risk of exposure to fine particles ($PM_{2.5}$)). In the case of air pollution, there is additional evidence linking exposure to both short- and long-term cognitive impairment. Through this mechanism, exposure might harm labour productivity or educational attainment (e.g. Clifford et al 2016). For other environmental risks, the evidence of harm or benefit is often more nuanced (see e.g. Brender et al. 2011 for the health impacts of living close to major pollution sources like industrial facilities, highways or waste management sites) or emerging (see e.g. Frumkin et al. 2017 and Gascon et al. 2017 for health impacts of access to green space and blue space).

While the associations between socioeconomic characteristics and environmental quality mostly defy broad generalisation, the importance of cities or the urban-rural divide is regularly highlighted and several studies find environmental inequality to be particularly an urban issue (e.g., Science for Environment Policy, 2016; Nieuwenhuijsen 2016; Fecht et al. 2015; Milojevic et al. 2017). Box 2.2 highlights some of the reasons why cities are relevant in this context.

Apart from any differences in exposure, evidence generally shows that disadvantaged groups are more *vulnerable* to exposure to environmental risks and disamenities, particularly with regards to health. They suffer worse consequences (i.e. are harmed more) than more advantaged groups even where the level of exposure is ostensibly the same (Science for Environment Policy, 2016). The evidence is particularly strong for exposure to air pollution. Evidence of these differences stretches over decades (see e.g., Winkelstein et al. 1967 for an early study on SES and air pollution). Possible explanations for this include the possibility that different groups that live in the same area may not actually be equally exposed to ambient outdoor air pollution because of variation in access to defensive measures such as air conditioning, better housing quality, better work environment, private transportation, etc. (e.g. Hajat et al. 2015); or that differences in vulnerability are caused by baseline health, other lifestyle factors or differences in quality of healthcare (e.g. Jans et al. 2018). Interestingly, this differential vulnerability suggests that improvements to environmental quality that benefit *everyone*, such as the reduction of background air pollution, may benefit disadvantaged groups the most (e.g. Milojevic et al. 2017). It also means that it may be more effective to focus on improving environmental conditions for poorer groups.

Box 2.2. The association between socioeconomic characteristics and environmental quality is particularly important in cities

Cities are relevant to discussions of environmental justice for several reasons:

- The majority of the world's population live in cities and urbanisation is continuing rapidly; it has been projected that 68% of the global population will live in urban areas by 2050 (UN 2018). Two-thirds of the OECD population already live in urban areas[4] (OECD 2013).
- Cities are known to exhibit dramatic spatial inequalities and segregation in terms of income and other socio-economic status (SES) characteristics (OECD, 2018c)[5]. Spatial inequality is a prerequisite for environmental inequality. Cities are also getting larger and evidence suggests cities become more unequal as they grow (e.g. Eeckhout et al. 2014).
- Cities are often more polluted. Air quality in particular is usually worse in cities. Some pollutants like NO_2 scale with population density (Lamsal et al. 2013) therefore the steep population density gradients around cities result in considerable spatial variation in pollutants. Spatial variation in environmental quality is also a prerequisite for environmental inequality.
- Noise pollution is generally worse in cities where the main noise emitters like roads, railways, airports, industry and households are concentrated.
- Access to certain amenities like green space is generally only limited in urban areas. Green space per capita declines as cities get larger (Fuller and Gaston 2009).
- Extreme heat is more of a problem in cities due to the urban heat island effect (Oke, 1967).

Cities may also be increasingly relevant because environmental policy decisions are increasingly made at the city level. Around two-thirds of metropolitan areas in the OECD now have a metropolitan governance body (OECD Metropolitan Governance Survey, 2014) and many spatially-targeted policies have been used in and around city centres in recent years such as an estimated 200 low-emission zones in European cities (Holman et al. 2015).

[4] Using the OECD *Functional Urban Area* definition.

[5] The degree to which the level of other types of inequality in a city and the variation in environmental quality are correlated does not seem to be have been directly addressed in the literature.

Key conclusions

The following conclusions emerge from the literature:

- The socio-economic distribution of exposure to environmental risk varies greatly by risk, status metric used, scale and location. However (and with exceptions), disadvantaged groups more often live in areas that are noisier, more polluted, more at risk of climate-related hazards and with less green space. These inequalities are likely to be more acute in cities. Furthermore, disadvantaged groups' health is also more vulnerable to some risks for a given exposure level, like air pollution (which is the environmental risk that causes the greatest harm).
- Identifying and responding to environmental inequality may present opportunities at different levels of government to reduce inequality more broadly, improve quality of life and help secure public support for the green transition. Aside from other benefits, it would almost certainly help to ensure that all groups equitably share the environmental improvements that accompany the transition to green growth.
- Future research could usefully be directed at cross-country studies that combine data on environmental quality and socio-economic status at a fine level of spatial disaggregation (e.g., census blocks, households or individuals) that allow comprehensive analysis at multiple scales (e.g. Fecht et al. 2015). The fact that such analyses span multiple countries allows the effect of (sub-)national environmental policies on inequality outcomes to be identified.

3. Evidence on the distributional effects of environmental policies

The environmental economics literature has a long tradition of analysing how different environmental policy instruments generate winners and losers by imposing different costs on individuals (Baumol and Oates 1988; Parry et al. 2006; Fullerton 2008). There is a broad consensus among economists that environmental policy should not generally be the tool for addressing distributional issues, since there are other more suitable mechanisms, although for political reasons it may be advantageous to address these issues within the context of the environmental policy (Johnstone and Serret, 2006).

There are three main types of possible distributional effects of environmental policy that are of relevance to individuals and households:[6]

1) The uneven distribution of policy-induced **environmental benefits** across individuals (e.g. the distribution of air pollution improvements across urban neighbourhoods): In cases when poor households tend to be exposed to more pollution, and suffer more of the harm of pollution to health, they are also more likely to benefit from improvements in environmental quality. This is a potentially important distributional channel, but as we shall see the actual distribution of benefits depends on targeted policy design.

2) The uneven distribution of **financial impacts** from environmental policy, including

    a) Higher prices of polluting products (e.g. an energy tax or a fuel economy standard raising the price of energy-intensive goods): Although poor households typically spend less, in absolute terms, on such goods and services than wealthy households, these expenditures represent a relatively higher share of their disposable income. However, besides household's income, the specific impact for a given household will also depend on household's expenditure structure (e.g. the share of energy-intensive goods in a household's consumption basket). Price increase is thus another potentially important distributional channel that will typically weigh disproportionally more on low-income households. However, these effects are temporary, at least to the extent that people substitute away polluting goods for cleaner ones.

    b) Capitalisation of the above two effects into asset prices (e.g. value of land and housing, financial capital).

3) The use of the fiscal revenues generated through environmental policy in **redistribution mechanisms** (targeted measures) to correct the possible regressive impacts. As we shall see below, the design of environmental policy instruments and any associated redistribution mechanisms will determine the overall distributional impact of a policy reform.

However, while the financial costs are more-or-less measurable, the non-market outcomes (e.g. environment-related health effects) are difficult to observe. "Due to this difficulty it is not generally known if most environmental policies are on net, progressive, regressive, or have no distributional effects" (Hsiang et al. 2018). Much of the literature is framed in

[6] There are other distributional effects of environmental policy that concern the factor markets and firms (for a review see e.g. Fullerton 2011) and these are discussed in another 2018 GGSD Forum scoping note on "environmental policy and firm competitiveness".

terms of policy instrument choice and design (i.e. how likely a given policy instrument is to generate a desirable distribution of impacts) and this is the angle adopted in this review.

Table 3.1 gives an overview of selected recent studies and a synthesis of the key findings follows next.

Evidence on the distributional effects environmental policies varies by instrument type. Regressive impacts can be neutralised, or even reversed, through targeted measures.

Taxes on energy and water use

Taxes on environmentally harmful products or activities are often too low to reflect the true costs of pollution to society. A frequent concern with raising such taxes is that they would translate into higher prices of goods and services with potentially regressive distributional impacts. For instance, higher energy taxes will be partially passed on the consumers of energy-intensive goods. And while richer households are affected more by an increase in taxes on domestic energy in absolute terms, such taxes are still likely to be *regressive* (in terms of energy expenditure as a share of income) if no additional policy measures are implemented (Flues and van Dender 2017).

However, the empirical evidence is more nuanced. In the specific case of transport fuel taxes, the common perception that such taxes are regressive is challenged by evidence suggesting neutral or only slightly regressive impact in richer countries and strong progressivity in middle- and low-income countries (e.g., Flues and Thomas 2015; Sterner 2012). The explanation put forward is that while in richer countries fuel is a normal good (quantity consumed rises in proportion to income), in poorer countries fuel is more of a luxury good that is consumed primarily by the well-off (Sterner 2012). Consequently, taxing transport fuel in poorer countries will impact essentially the richer households. Most importantly, the literature is clear in that regressive outcomes can be reversed or alleviated through targeted measures (transfer payments, revenue recycling or tax swapping), achieving an overall progressive distribution of impacts (e.g., Flues and van Dender 2017; Sterner 2012; Heindl and Löschel 2014). Poorer households can thus benefit from well-designed energy tax reforms.[7]

This finding is confirmed by Flues and van Dender (2017) in a simulation conducted on real-world data from OECD Countries. They show that lump-sum transfers can mitigate any regressive impacts, and income-tested transfers can even result in *progressive* impacts (i.e. energy expenditure as a share of income increases stronger for richer households) of the tax reform overall (see Box 3.1).

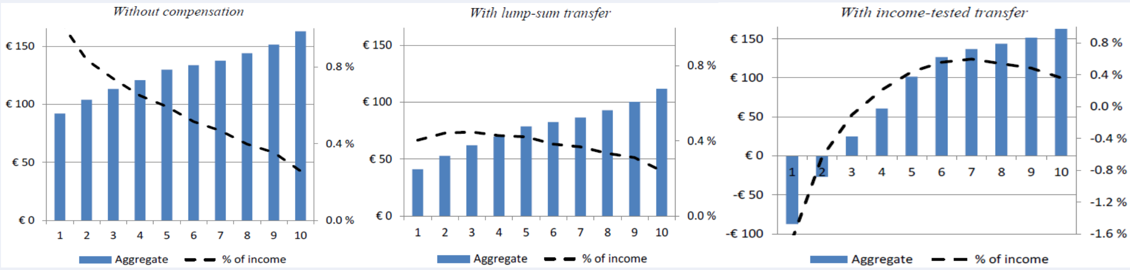
Identical conclusions can be drawn from the literature on residential water charges. For instance, a large cross-country household survey found that, in the absence of any cash transfers, low-income households will be most adversely affected by increases in volumetric water charges as they spend proportionately more than twice as much on residential water use than high-income households (OECD, 2011b). This confirms earlier studies that found that the burden of water charges can be up to four times greater for the lowest decile income group when compared to the average burden across all households (OECD, 2003). Here again, the recommended course of action is to implement full-cost

[7] However, some low- and middle-income countries face the challenge of putting in place effective transfer systems which reach all the poor.

water pricing, coupled with financial assistance to low-income households via transfer payments (OECD, 2011b).

Box 3.1. Rich households' energy tax contributions increase the most after the tax reform

Simulated energy tax reform in 20 OECD countries, average tax difference per household



Source: Flues and van Dender (2017, p.31)

In the above Figure, panel 1 shows that richer households contribute more to energy tax revenues in absolute terms than poorer households in response to a simulated energy tax reform drawing on detailed household-level microdata from 20 OECD countries.

Panel 2 shows that introducing *lump sum transfers* (every household receives the same absolute relief per person) leads to a distribution of impacts of the reform that are still somewhat regressive but considerably less so than without any transfers. Poor households still contribute more on average than without the reform, but substantially less than richer households.

Panel 3 shows that introducing *income-tested cash transfers* generates a progressive incidence of the reform, with households in the lowest two income deciles better off on average after the reform. The income-tested cash transfer ensures that the impact of the reform falls more on the richer households both in absolute and in relative terms.

This exercise demonstrates that poor households can benefit from an appropriately designed energy tax reform.

Standards for ambient air quality and building energy performance

There is a broad consensus in the literature that taxes are often a more cost-efficient and more environmentally effective means of mitigating emissions of harmful substances or reducing energy and water consumption. It is thus striking that technology and performance standards remain a common way of regulating environmental externalities. This is partly because the distributional impacts of standards are perceived to be less regressive as those of taxes, although recent empirical evidence finds exactly the opposite that environmental standards can be more regressive than taxes (Fullerton and Muehlegger 2017). This mistaken perception is largely due to the costs of standards being mostly hidden while the costs of environmentally related taxes are explicit (Johnstone and Serret 2006). Moreover, the benefits of standards are defined upfront (although subject to compliance) while those of taxes depend on market responses.[8]

It is precisely due to the latter (uncertainty over environmental damages) that performance standards are a common way of regulating local air quality. Recent evidence shows that the benefits of such policies may be progressive if they provide incentives to target abatement efforts at pollution hotspots. For instance, following the 1990 US Clean Air Act Amendments local regulators had incentives to target areas with the highest pollutant concentrations, leading to air quality improvements that were highly localised. As houses in the most polluted areas tend to be owned by relatively lower-income households on average, these homeowners enjoyed the largest improvements in air quality and hence the highest relative house price appreciation. The poorest households received annual benefits from the programme that were two times higher relative to their incomes than the wealthiest households (Bento et al. 2015).

Another instance when the introduction of performance standard may be suitable is the presence of informational market failures (e.g. potential house buyers cannot perfectly observe building quality) and split incentives (e.g. landlord-tenant[9]). For instance, building energy codes are a common measure in many countries intended to reduce energy consumption by setting minimum efficiency requirements for new constructions. Their introduction is sometimes motivated also by distributional concerns (i.e. more preferable distributional outcome than would be feasible with energy taxes). However, the empirical literature does not support this conjecture. Recent evidence suggests that building performance standards lead to more distortions for lower-income households and regressive impacts on household wealth – for instance, because stricter building codes lead to smaller dwellings for lower-income households and only rather limited induced energy use savings (Bruegge et al. 2018).[10]

Subsidies for consumption of water, energy and parking space

Another widely held perception is that water and energy consumption subsidies benefit the poor, for instance, the increasing block tariffs (IBT) that are commonly applied by water utilities charge higher rates with increasing consumption. It is often believed that such a tariff structure not only provides low-income households access to water at affordable price, while at the same time providing water-saving incentives and allowing cost recovery from consumers who use more water. The implicit assumption is that all households have access to the water network and that low-income households use less water than high-income households. However, the empirical literature challenges this view showing that the de facto subsidies delivered through the water tariff are poorly targeted and largely regressive. For instance, Fuente et al. (2016) show that the IBT implemented in Kenya does not target subsidies to low-income households effectively and instead the higher-income customers receive a disproportionate share of subsidies. They put forward three possible reasons for the regressive incidence of the water tariff structure in low- and middle-income countries: (i) poor households lack a piped connection, effectively excluding them from

[8] In addition, standards do not raise any public revenue that could be used to mitigate or reverse the potential regressive effects of standards.

[9] E.g. owners of rental property have an incentive to underinvest in energy- and water-efficient facilities because such investments largely benefit the tenants who typically pay the bill for utilities.

[10] Recent empirical evidence questions the effectiveness of building energy codes in reducing energy use more generally (e.g. Levinson 2016; Davis et al 2018).

subsidies provided through low-priced water delivery, (ii) multiple households may share a connection served with a single meter exposing them to a higher tariff than if they each had individual metered connection, and finally (iii) the empirical correlation between household income and water use is lower than what is commonly assumed and the tariff is generally too low to cover the service costs. The latter implies that subsidies are poorly targeted even among households with a piped connection.

Explicit or implicit subsidies on the consumption of fossil fuels, or energy use more broadly, continue to be ubiquitous in many countries. This is despite the environmental damages of such subsidies (e.g. local air pollution, GHG emissions), their financial costs (increased burden on public budgets) and lower well-being and living standards (e.g. premature deaths, reduced labour productivity due to human health impacts). Vested interests combined with inadequate democratic representation, and concerns over the distributional impact on poorer households often stand in the way of their removal. Durand-Lasserve et al. (2015) show that a phase-out of all energy consumption subsidies in Indonesia would result not only in reduced emissions but also in gains of real GDP and consumer welfare. The overall distributional performance of the reform would depend on the particular redistribution scheme applied: Cash transfers (direct payments to households, such as lump sum or means-tested transfers) are the best way to make a phase-out more efficient and equitable because they align prices with costs, and they make the reform more attractive for poorer households and reduce poverty. On the contrary, redistribution via income tax relief which is proportional to household income is more beneficial to higher income households and increases poverty.

Free on-street parking is another form of consumption subsidy with possible adverse distributional impacts, especially in low- and middle-income countries, because it unequally distributes the value of public land to car-owning (higher-income) households. For instance, in a pilot programme from Beijing only a third of households owned a car and the average income of car-owning households was almost three times the income of carless households (Shoup et al. 2017). The study shows how charging market prices for on-street parking allows not only to generate environmental benefits (reduce over-crowding in streets) but also to more equally distribute the value of public land used for on-street parking if the revenue raised is used to finance local public services. Russo et al. (2018) draw similar conclusions from a study of a number of OECD cities. They suggest that most of the measures directed at more efficient and environmentally sustainable parking policies (e.g. appropriate pricing of on-street parking and residential parking permits, removing minimum parking restrictions for new residential and office buildings, and reconsidering exemptions of employer-paid parking from income taxation) would also likely lead to direct distributional benefits. In the cases where the direct effects of these changes on vulnerable population groups are negative, these groups can be compensated through targeted complementary measures.

Compensation schemes for voluntary provision of ecosystem services

The costs of policies to conserve and sustainably use biodiversity are generally borne by the populations living in the area where these policies are implemented. As the majority of biodiversity is found in developing countries, they are often burdened with the costs of biodiversity policies while a significant proportion of the benefits accrue worldwide (OECD, 2012b). International financing mechanisms such as payments for ecosystem services (PES) are an increasingly common instrument to incentivise voluntary land conservation, particularly forests. By offering conditional cash transfers to landowners they increase the private returns to forestry and thus reduce the difference between private and social values of forest. In addition to the environmental benefits of PES programmes, there is increasingly interest in their associated developmental and social benefits such as poverty alleviation. The empirical literature suggests that PES are likely to have "positive but modest livelihood impacts on ecosystem service suppliers" (Liu and Kontoleon, 2018). To achieve stronger social outcomes, programme managers face difficult trade-offs against environmental effectiveness "illustrating the difficulty of meeting multiple policy goals with one tool" (Alix-Garcia et al. 2015).

Moreover, there are broader concerns about using such programmes to achieve poverty reduction objectives. This is because schemes involving ownership-based eligibility requirements (like payments for ecosystem services or subsidies for renewable energy generation) will exclude the disenfranchised who do not own land, homes or other assets.

Table 3.1. Selected empirical evidence on the distributional effects (benefits and costs, financial and non-financial) of environmental policies

Items have been grouped by policy instrument type

Study	Scope	Key conclusions about incidence
	Taxes and charges	
Flues and van Dender (2017)	Domestic energy use in 20 OECD countries, drawing on household-level microdata	A simulated energy tax reform that increases taxes on electricity and heating fuels (achieving uniform marginal tax rates across fuels and countries) can improve energy affordability if part of the additional tax revenue is redistributed back to households using an income-tested cash transfer.
Flues and Thomas (2015)	Energy use in 21 OECD countries	*Direct* distributional effects (i.e. before or without revenue recycling) of existing energy taxes differ by energy carrier: Taxes on transport fuels are often proportional in high-income countries and progressive in middle-income countries. Taxes on heating fuels and electricity are found to be regressive in many cases but the overall tax contribution on these goods is low, with little effect on the overall distributional impact of the tax systems in the countries analysed.
Sterner (2012), Slunge and Sterner (2009)	Transport fuel use, drawing on case studies from Asia and Africa	Finding that while there may be slightly regressive *direct* effects (i.e. in the absence of revenue recycling) in some high-income countries, as a general rule, fuel taxation is a progressive policy particularly in low income countries. Rich countries can correct for regressivity by cutting back on other taxes that adversely affect poor people, or by spending more on services for the poor.
Heindl and Löschel (2014)	Energy use, review of over 120 studies	Conclusion that if all or a sufficient part of revenue from energy taxes is handed back to the consumer, it is

		possible to avoid negative distributional effects, and even decrease poverty and deprivation.
OECD (2011b)	Residential water use, drawing on a household survey implemented in 10 OECD countries	Low-income households are likely to be most adversely affected by increases in volumetric water charges as they spend proportionately more than twice as much on residential water use than high-income households. Full-cost water pricing should be coupled with assistance to low-income households in the form of a low or zero fixed fee, or via transfer payments.
	Subsidies	
Fuente et al. (2016)	Water tariffs in Kenya, drawing on household-level microdata	High-income residential and non-residential customers receive a disproportionate share of subsidies.
Fuente et al. (2016)	Water subsidies, review of 21 studies from Latin America, Africa, Asia and Europe	The subsidies delivered through water utility tariffs are poorly targeted and largely regressive.
Durand-Lasserve et al. (2015)	Energy subsidy reform, application of a CGE model to Indonesia.	A phase-out of all fossil fuel and electricity consumption subsidies, combined with redistribution through direct payments to households (cash transfers) would reduce poverty and generate the highest aggregate gains in real GDP and consumer welfare. Redistribution via labour tax relief is, on the contrary, more beneficial to higher income households and increases poverty.
Shoup et al. (2017)	Parking policy reform in Beijing	Free on-street parking unequally distributes the value of public land to car owners and provides a de facto subsidy to car-owning households. Introduction of on-street parking charges provides public revenue to finance local public services, generating relatively more benefits for low-income households that are less likely to own a car.
	Compensation schemes for voluntary provision of ecosystem services	
Liu and Kontoleon (2018)	Payments for ecosystem services, meta-analysis	Finding that PES programmes are likely to have positive but modest livelihood impacts on ecosystem service suppliers.
Alix-Garcia et al. (2015)	Payments for ecosystem services in Mexico, using household survey	Evidence of small progressive impacts. The programme reduces the expected land cover loss by 40–51 percent and modestly alleviates poverty.

Grieg-Gran et al. (2005)	Payments for ecosystem services in Latin America	Distribution of benefits depends mostly on the eligibility rules (e.g. hectare caps to limit payments to wealthy landowners, formal land ownership which may limit access to the programme for the poor) and pro-poor measures (e.g. reduced smallholders' transaction costs).
Performance standards		
Bento et al. (2015)	Air quality in the US	Lower-income homeowners benefited more from the 1990 US Clean Air Act Amendments, as these were the homeowners located in areas that experienced the largest improvements in air quality and hence the highest relative house price appreciation.
Bruegge et al. (2018)	Building energy codes in the US	Lower-income households suffer the largest distortions to home characteristics (home area and the number of bedrooms) and a decline in home value, while high-income households experience very small distortions in these characteristics and an overall increase in home value.
Fullerton and Muehlegger (2017)	Review of energy and carbon-related mandates	Conclusion that environmental standards (or mandates) can be more regressive than taxes.

Key conclusions

The following conclusions emerge from the literature:

- Environmental tax reforms that include measures to neutralise regressive impacts on vulnerable households can deliver environmental improvements cost-efficiently while avoiding negative distributional impacts. Depending on the design, they can even reduce poverty and income inequality.
- Similarly, removal of environmentally harmful subsidies can achieve neutral or progressive impact if accompanied by compensatory measures targeted at the poor (e.g. via existing social welfare systems).
- The common belief that taxes have larger direct (i.e. before revenue recycling) regressive impacts than performance standards is due to the greater 'visibility' of the costs of taxes compared to standards. In addition, standards do not raise any public revenue that could be used to mitigate or reverse the potential regressive effects of standards.
- Evidence that in low- and middle-income countries environmental taxes often have neutral or progressive impacts because the taxed goods tend to be predominantly consumed by rich households.
- More generally, environmental policies can strengthen progressive outcomes if they provide incentives to target mitigation efforts in ways that deliver benefits primarily to the poorer households. However, such strategy is likely to generate trade-offs with environmental effectiveness and cost-efficiency.

4. Conclusions and areas for future work

Current state of knowledge about the distribution of environmental quality and areas for future work

The socio-economic distribution of exposure to environmental risk varies greatly by risk, status metric used, scale and location. With many exceptions, evidence suggests disadvantaged groups more often live in areas that are noisier, more polluted, more at risk of climate-related hazards and with less green space. Such environmental inequality is likely to be more acute in cities. Identifying and responding to environmental inequality may present opportunities at different levels of government to reduce inequality more broadly, improve quality of life and help secure public support for the green transition

Empirical assessment is necessary to identify the existence and severity of any environmental inequalities on a case-by-case basis.

While there are places where environmental inequality is more likely to be a concern – such as large cities, places that are more segregated or places where environmental quality is highly varied, exceptions are common – therefore, empirical analysis is required to determine if less advantaged groups suffer a disproportionate burden in a specific area.

Better data would allow a more comprehensive understanding of the patterns and drivers of inequality.

Data on some facets of environmental quality or risks such as air pollution is improving. Information on spatial distribution of socioeconomic status is also often available. However the empirical challenges of comparing the levels of environmental inequality between countries to understand the potential role of policies requires a level of data comprehensiveness and harmonisation that does not yet exist.

Future research could usefully be directed at cross-country studies that combine data on environmental quality and socio-economic status at a fine level of spatial disaggregation that allow comprehensive analysis at multiple scales (e.g. Fecht et al. 2015). The fact that such analyses span multiple countries allows the effect of environmental policies on inequality outcomes to be isolated.

Best practices to address distributional impacts of environmental policies

Policies that lead to across-the-board environmental improvement seem unlikely to raise distributional concerns, and may be progressive.

Given the evidence that poorer groups seem more vulnerable to environmental risks, across-the-board reductions in risk are more likely to be progressive (i.e. benefit poorer households more) than regressive (however the progressivity is probably modest and fiscal distributional concerns may remain). This conclusion does not necessarily apply to spatially-targeted policies where, by design, some people benefit more than others such as low-emission zones. The distributional impacts of these kinds of policies may warrant further study.

Targeting mitigation incentives at areas where vulnerability to pollution is highest can help augment the progressivity of a policy.

Insofar the poor tend to be located in areas with higher pollution, then targeting the most polluted areas will provide more environmental benefits to the poor. Moreover, these are also among the most vulnerable socioeconomic groups due to a greater risk of lost earnings due to health impacts of pollution and generally lower baseline health. For instance, the United States Clean Air Act Amendments provide incentives to abate pollution in areas where pollution levels are highest, leading to relatively larger environmental benefits for the poorer neighbourhoods (Bento et al. 2015).

Risks of exclusion need to be addressed

One mechanism for policies to generate progressive outcomes involves a greater capitalisation of environmental benefits in asset prices (e.g. land and home values) owned by the poorer households. However, the poorest households that do not own such assets will be left out. Policies involving ownership-based eligibility requirements like payments for ecosystem services or subsidies for renewable energy generation require particular care because the disenfranchised who do not own land, homes or other assets lose out.

Revenue recycling schemes play a key role

Raising the cost of pollution is a prerequisite of any effective policy, such as energy tax reforms or the removal of harmful energy consumption subsidies. Any concerns over regressive impacts of such policy reforms can be mitigated and fully compensated through effective policy design with revenue recycling as the most important aspect (Heindl and Löschel 2014). Such corrective measures should be an intrinsic part in the design of environmental policy reforms.

The particular type of a redistribution scheme ultimately matters in determining the overall distributional performance of the reform. Cash transfers directed at deprived households, spending on public services directed at the poor, or cutting back on other taxes that adversely affect poor people, can make the reform more attractive for poorer households and reduce poverty. Importantly, means-tested redistribution is preferred to lump-sum transfers and benefits because it allows achieving more progressive distributional outcomes. In contrast, mechanisms that compensate households via payments proportional to labour income (such as income tax relief) are more beneficial to higher income households and increase poverty (e.g. Durand-Lasserve et al. 2015; Flues and van Dender 2017; Sterner 2011). Well-functioning social welfare systems are needed to effectively achieve the desired distributional outcomes.

In sum, ensuring that outcomes of environmental policy reforms are more inclusive matters for at least two reasons. First, it can help overcome some of the latent resistance to (environmental) policy reforms, increasing policy effectiveness. Second, climate and environmental-related factors can aggravate existing health risks particularly for vulnerable populations. Improving environmental quality for the poor and reducing their vulnerability to climate risks can help addressing some of the other types of inequality and even reduce overall levels of poverty. However, **environmental policy-makers should not lose sight of their principal objective which is to reduce environmental pressures, therefore, it is imperative that the distributional concerns are addressed in a manner that does not undermine the incentives to reduce pollution or use resources sustainably.**

Questions for discussion

- What are some of the most pervasive examples of environmental damages that are distributed unequally across populations? Is environmental inequality best regarded as primarily a city-level problem, to be addressed by municipalities?
- Is there a ‘virtuous circle’ of equality where better environmental equality helps improve other types of equality which in turn helps further improve environmental equality?
- Could environmental factors be a key driver for the distribution of wealth?
- Could environmental damages that are incurred today reinforce disparities within future generations?
- What are some of the most pervasive examples of benefits from environmental policies that are distributed unequally across populations?
- How can policy prevent potential undesirable impacts on the distribution of environmental quality? How can regressive financial impacts of environmental policies be mitigated or avoided?
- What opportunities are there to promote inclusivity via environmental policies? To what extent can (or should) environmental policy be used to solve the wider issues of inequality?
- What are the barriers for governments to more often use targeted redistribution mechanisms (revenue recycling) in order to avoid negative distributional consequences of environmental policy reforms? Are social benefits systems working well, or is there a need for reform when environmental policy become more stringent?

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