Waste Incinerators Undermine Clean Energy Goals

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Abstract
A national clean energy standard, modeled upon existing state-level Renewable Portfolio Standards, has been proposed to decarbonize the U.S. electric grid. Most such state policies include municipal solid waste incineration as a form of "renewable" energy, despite incineration's prominent role in perpetuating environmental injustice. This study finds that incinerators emit more greenhouse gas emissions per unit of electricity produced than any other power source. They also emit more criteria air pollutants than replacement sources of energy. Incineration's inclusion in "renewable" or "clean" energy standards is thus counterproductive, as they also divert more than $40 million in subsidies annually from cleaner energy sources. As the electric grid decarbonizes, these disparities will only grow. With most U.S. incinerators nearing their end of life, policy choices about their eligibility for subsidies may well decide whether they shut down or undertake expensive capital improvements to continue operating. A rapid shutdown of existing incinerators would help decarbonize the electric grid and reduce criteria air pollution, particularly in environmental justice communities.

Keywords
Waste incineration
Renewable energy
Greenhouse gas mitigation
Environmental justice

1. Introduction

As the United States prepares to rejoin the Paris Agreement, a variety of greenhouse gas (GHG) mitigation policies have been proposed to enable it to reach the goal of zero net emissions by mid-century. The electric grid is widely considered the easiest and quickest sector to decarbonize and the Biden administration has pledged to achieve a climate-neutral electric grid by 2035 ("The Biden Plan to Build a Modern, Sustainable Infrastructure and an Equitable Clean Energy Future," n.d.). A leading candidate for achieving this goal is a national clean electricity standard, modeled upon existing state-level policies (e.g., U.S. House Select Committee on the Climate Crisis, 2020). These policies, generally known as Renewable Portfolio Standards (RPS), require electric grid operators to buy a minimum percentage of electricity from renewable energy sources. Renewable Energy Credits (RECs) serve to track the amount of renewable electricity produced; as RECs are separable from the electricity supply, parallel REC markets have been created, and the surrender of RECs to a regulatory authority does not necessarily correspond to the generation of renewable electricity within that state (Mack et al., 2011). Moreover, there is no consistent definition of "renewable": each state determines which energy sources qualify for its RPS. The GHG emissions impact of existing RPS policies is debated: most researchers find significant reductions while a few find minimal effects (Barbose et al., 2016; Upton & Snyder, 2017). The inclusion of energy sources with large carbon footprints within the definition of "renewable" may be one reason for the policies' weak mitigation impact.

Municipal solid waste incinerators (MSWI or "incinerators") are one of the more controversial technologies subsidized by RPS programs. Claims to renewability hinge upon the biomass component of solid waste, which includes food waste, paper, and cardboard, being considered climate neutral when combusted. However, the rapid increase in plastic waste – which is almost entirely derived from fossil fuels – weakens this claim (Pratt & Lenaghan, 2020). Proponents also argue that incinerators avoid methane emissions from the landfilling of the putrescible component of solid waste and recover energy from materials that would otherwise be wasted (Michaels, 2009); these analyses, while numerous, are generally of poor quality (Astrup et al., 2015). Other strategies, such as composting and landfill gas collection, also avoid methane emissions and have been shown to have lower GHG emissions than incineration, as does recycling; these strategies compete with incineration for fractions of the municipal waste stream (Morris et al., 2013; Powell et al., 2016; USEPA, 2006, 2009). In any case, it is not clear that waste management questions are germane to a definition of renewable or low-carbon electricity. The industry has increasingly argued that incineration is a source of "clean" or low-carbon energy, de-emphasizing its role in waste management (e.g., Brown, 2014; Grosso et al., 2010; Michaels, 2009). In this paper, we focus on the role of incineration as an energy source rather than compare it to other waste management strategies.

Incinerators have frequently attracted the ire of their host communities. In addition to greenhouse gas emissions, incinerators are major sources of toxic air emissions including dioxins, lead, mercury, nitrogen and sulfur oxides, and particulates (Baptista & Perovich, 2019). In some cities, the incinerator is the single largest source of criteria air pollutants (McAnulty, 2019). Incinerators are primarily (79%) located in communities of color and low-income communities whose residents are subject to multiple, cumulative health impacts (Baptista & Perovich, 2019). Their presence has served to attract other polluting industries and requires large volumes of heavy truck traffic, with its ensuing emissions (Baptista & Perovich, 2019). While permitting standards consider the facility in isolation, most of their host communities struggle with multiple environmental health stressors, which exacerbate incinerators' impact. As such, environmental justice movements have frequently targeted incinerators for closure (Behrsin, 2019).

Nevertheless, incineration is included in 26 of the 42 state-level RPS programs (Table A1).[1] Industry analysts claim that the subsidy afforded by RPS programs is critical to the expansion and maintenance of the country's incinerator fleet, which would otherwise be uneconomical to operate (Baptista & Perovich, 2019; Behrsin, 2020). Most U.S. incinerators are nearing the end of their expected operating lives and would require major capital investments to continue operations (Baptista & Perovich, 2019). The decisions to decommission or refurbish incinerators – or to build new ones – may hinge on incineration's inclusion in federal and state RPS programs.

In this paper, we assess incineration's impact on U.S. greenhouse gas emissions by calculating incinerators' excess emissions. We define a plant's excess emissions as the plant's emissions minus the emissions that would be generated by replacement power sources. Previous analyses have found that the climate impact of waste-to-energy is heavily dependent on assumptions made

[1] 21 states, the District of Columbia, Guam, the Northern Marianas Islands, Puerto Rico, and the U.S. Virgin Islands, which we collectively refer to as "states".

about replacement electricity (Astrup et al., 2015; Morris et al., 2013; Schott et al., 2016; Smith et al., 2001). In constructing likely scenarios for replacement electricity, several factors need to be considered: geographic area, replacement ratio, and marginal vs. average emissions.

The task of matching electricity generation to demand falls to 73 balancing authorities, each with authority over a section of the U.S. electric grid. However, replacement sources of electricity are not necessarily located within the geographic area of a balancing authority, due to large-scale transfers of electricity between balancing authorities. Accounting for these transfers and their associated emissions, is impractical, if not impossible (Ryan et al., 2016). The Environmental Protection Agency (EPA) has defined subregions (an intermediate geographic area between balancing authorities and regions defined by the North America Electric Reliability Corporation) so as to minimize the import and export of electricity across boundaries (USEPA, 2020b). Subregional analysis allows the closest match between electricity demand and associated emissions.

In analyzing the emissions impact of adding or removing an energy source to the electric grid, it is standard practice to assume one-to-one replacement by other energy sources. In other words, if incinerators were to shut down, their electricity production would have to be compensated by an equal quantity of increased generation from other sources. However, an examination of 50 years of international panel data found that this is often not the case: in practice, alternative energy sources displace on average only 10% of their electricity output (York, 2012). For biomass and waste incineration, no displacement is discernible. Similar dynamics have been observed with biofuels (Hochman et al., 2010). Reasons for the lack of displacement may include lock-in to the existing electric system, the political and economic power of the fossil fuel industry, elasticity effects, and simply the relatively small electricity output of incinerators.

2. Data and methods

We used annual emissions and electricity generation data from the U.S. Environmental Protection Agency's 2018 Emissions and Generation Resource Integrated Database version 2 (eGRID), a database commonly used for power system analysis (USEPA, 2020a). For combined heat and power (CHP; also known as cogeneration) plants, we used eGRID's allocation factor to apportion emissions between electricity and heat production. We report fossil carbon dioxide (CO_2), biogenic CO_2, and the gases with high global warming potential (GWP) – methane (CH_4) and nitrous oxide (N_2O) – separately. In addition, we conduct similar analyses for sulfur dioxide (SO_2) and nitrogen oxides (NO_x), non-greenhouse gases included in the eGRID database that are important contributors to poor air quality and acid rain. Mercury emissions are also included in eGRID but, as of 2018, data coverage is too sparse for meaningful analysis (USEPA, 2020b). See Appendix for additional methodological details.

To evaluate the excess, or net, emissions from incinerators in 2018, we deducted the emissions associated with replacement energy – both heat and electricity – from each incinerator's emissions. We assumed that each CHP incinerator would be replaced by a natural gas facility of median emissions intensity with an overall energy output equal to the replaced incinerator's (i.e., 100% replacement). For electricity-only incinerators, we evaluated three scenarios, with replacement coefficients of 0%, 50%, and 100%, to capture the range of possible replacement effects. We use the EPA's subregion as the geographic unit of analysis (Figure A1).

Incinerators typically operate 24 hours a day, throughout the year; compared to time-variant generators such as wind, solar, and natural gas "peaker" plants, their electricity output is relatively constant. As such, they are likely to be replaced by sources that reflect the full mix of grid sources, and we use mean rather than marginal emissions intensity to calculate replacement emissions (Ryan et al., 2018).

Mean emissions intensity is changing over time, as natural gas and renewables replace coal; such changes will need to accelerate if the U.S. is to meet Paris Agreement goals of net zero emissions by 2050. We construct two future decarbonization scenarios ("Business as unusual," 2017). In the "No policy" scenario, each subregion of the grid decarbonizes at 2.3% annually, the rate that best fits the U.S. Power Sector Carbon Index from 2001-2019 (Schivley et al., 2018). In the "Paris" scenario, we combine emissions data from Grubert's (2020) reference scenario for plant retirements with electricity generation data from the Energy Information Agency's reference projections (U.S. Energy Information Administration, 2020). This results in an annual 9.9% decarbonization rate for fossil CO_2, which we apply to all emissions categories.

Throughout the analysis, we have used conservative assumptions – i.e., assumptions that would tend to understate the emissions impact of incinerators. These include the use of 100% replacement of both electricity and heat; the use of 100-year GWPs; and assuming no decarbonization in the CHP sector through 2050. As such, our estimates of excess emissions from incinerators should be seen as a lower bound.

3. Results

Of the nation's 76 operating incinerators, 69 were analyzed; of these, 56 are electricity-only plants and 13 are CHP facilities. Incinerators are the most emissions-intensive form of power generation: per unit of electricity produced, incinerators emit 1.7 times as much GHGs, 4.8 times as much NO_x but only 0.4 times as much SO_2 as coal, the next most polluting fuel. Compared to the national grid average, incinerators emit 3.8 times as much GHGs, 14 times as much NO_x, and 1.3 times as much SO_2 (Figures 1, 2, and A2; Table 1). Coal-fired plants emit 19% more fossil CO_2 than incinerators but negligible biogenic emissions. Biomass plants emit low levels of fossil CO_2 and 17% less biogenic CO_2 than waste incinerators. Waste incinerators stand out as the only generation source that emits large quantities of both fossil and biogenic emissions for each unit of electricity produced.

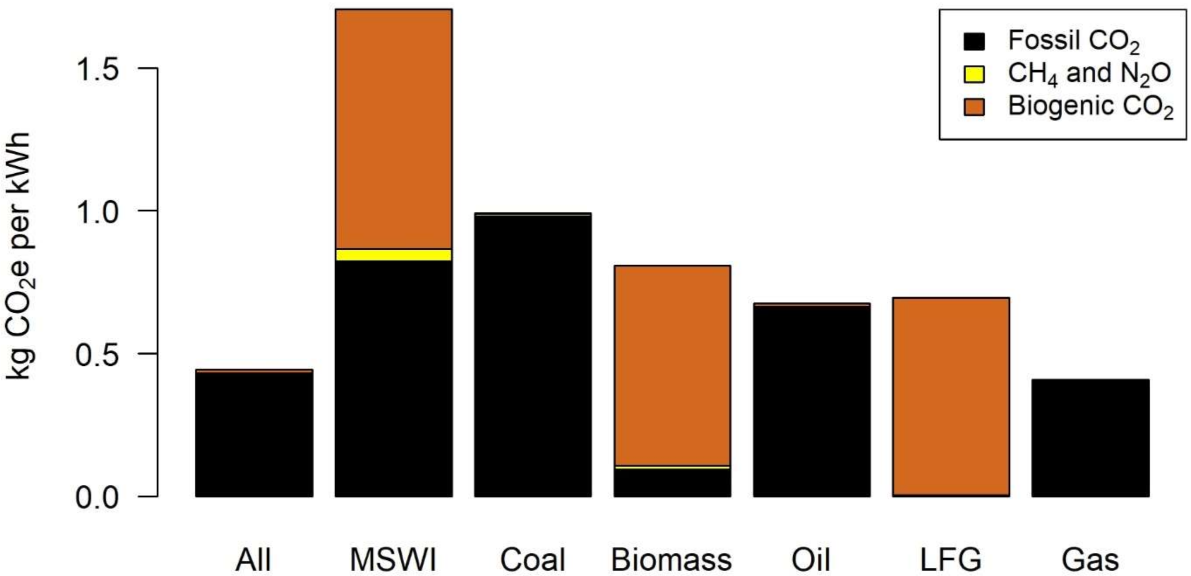

Figure 1: Generation-weighted mean national GHG emissions intensity by major fuel type for electricity. "MSWI" is municipal solid waste incineration, "LFG" is landfill gas, and "Gas" is natural gas.

The electricity mix varies widely by subregion. In 11 of the 15 subregions containing incinerators, incineration is the most GHG-intensive source of electricity (Figure A3). Exceptions include NPCC New York City/Westchester, where oil has a very high emissions intensity, on par with incineration; and RFC Michigan, RFC West, and SPP South, where incineration emissions intensities are anomalously low. These low values may be due to issues in the calculation of CHP emissions (see section 4.3).

	All	**MSWI**	**Coal**	**Biomass**	**Oil**	**LFG**	**Gas**
Fossil CO_2	428.6	822.9	981.6	95	660.2	4.4	406.2
$CH_4 + N_2O$	2.9	42.6	8.5	13.4	3.7	0.8	0.5
Biogenic CO_2	12.7	841.7	1.0	699.9	11.9	689.8	1.2
GHGs	444.2	1707.2	991.1	808.2	675.8	695.0	407.9
NO_x	10.9	152.9	31.9	51.7	14.6	2.6	1.6
SO_2	0.3	0.4	1.0	0.9	2.2	0.2	0.0
Plants	8590	69	336	264	662	338	1603
Electricity	100%	0.3%	28.8%	1.0%	0.6%	0.2%	33.5%

Table 1: Generation-weighted national electricity emissions intensity by major fuel type in grams (CO_2e for CH_4, N_2O) per kWh. "GHGs" is the sum of all greenhouse gases. "Plants" is the number of facilities. "Electricity" is the percentage of grid power supplied, including electricity from cogeneration (CHP) facilities. Percentages do not total 100% because 34.9% of electricity is generated from non-combustion sources not included in the table.

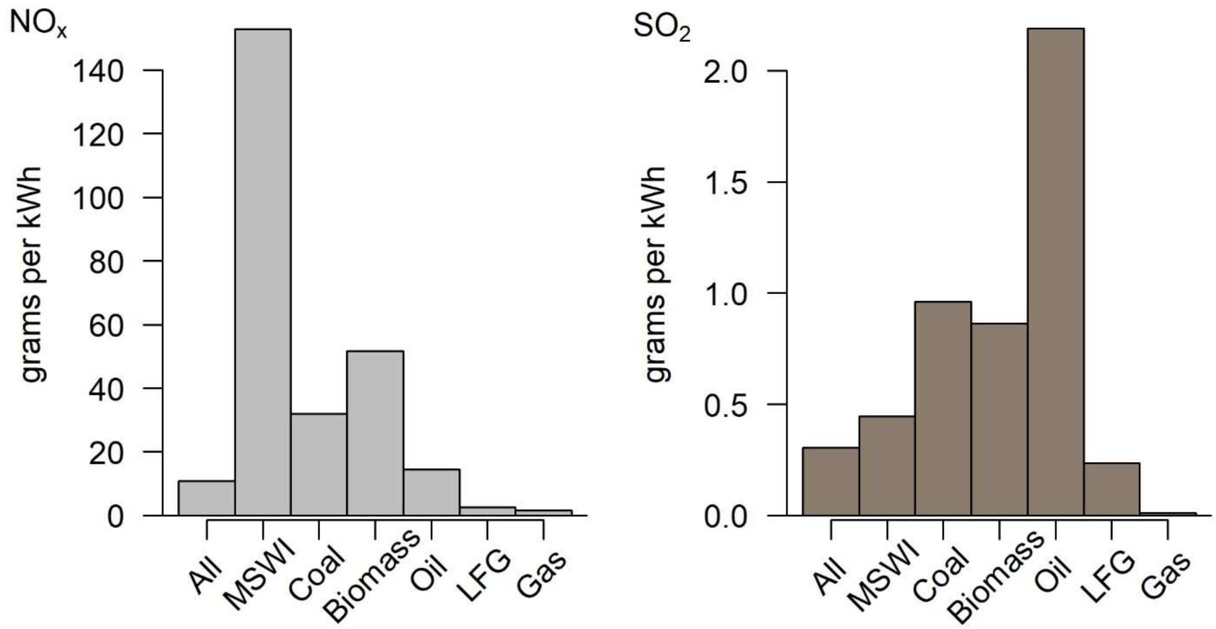

Figure 2: Generation-weighted mean national NO_x and SO_2 emissions intensity by major fuel type for electricity. "MSWI" is municipal solid waste incineration, "LFG" is landfill gas, and "Gas" is natural gas.

In 2018, with one exception, every incinerator produced excess emissions of each gas (fossil CO_2, N_2O and CH_4, biogenic CO_2, NO_x) other than SO_2 (Figures 3, 4). This is true regardless of the rate of energy replacement assumed. The one exception was the Pittsfield facility, which primarily (95.2%) produces heat rather than electricity. Its fossil CO_2 emissions were 4.7% less than a natural gas replacement; but this difference is more than made up for by its higher N_2O and CH_4 emissions.

Nationally, incinerators also produced excess SO_2 emissions in 2018, but substantial variability in plant performance and replacement SO_2 intensity produced wide variations in individual excess emissions rates. Notably, Hawaii has very high SO_2 grid intensity due to its reliance on oil (69% of generation) and thus its incinerator produces substantially less SO_2 emissions than ostensible replacement sources (-1111 tonnes in 2018 under 100% energy replacement). On the other hand, just nine heavy emitters produce all the national excess emissions. Removing these ten outliers would reduce incinerators' excess emissions nationally by 75%.

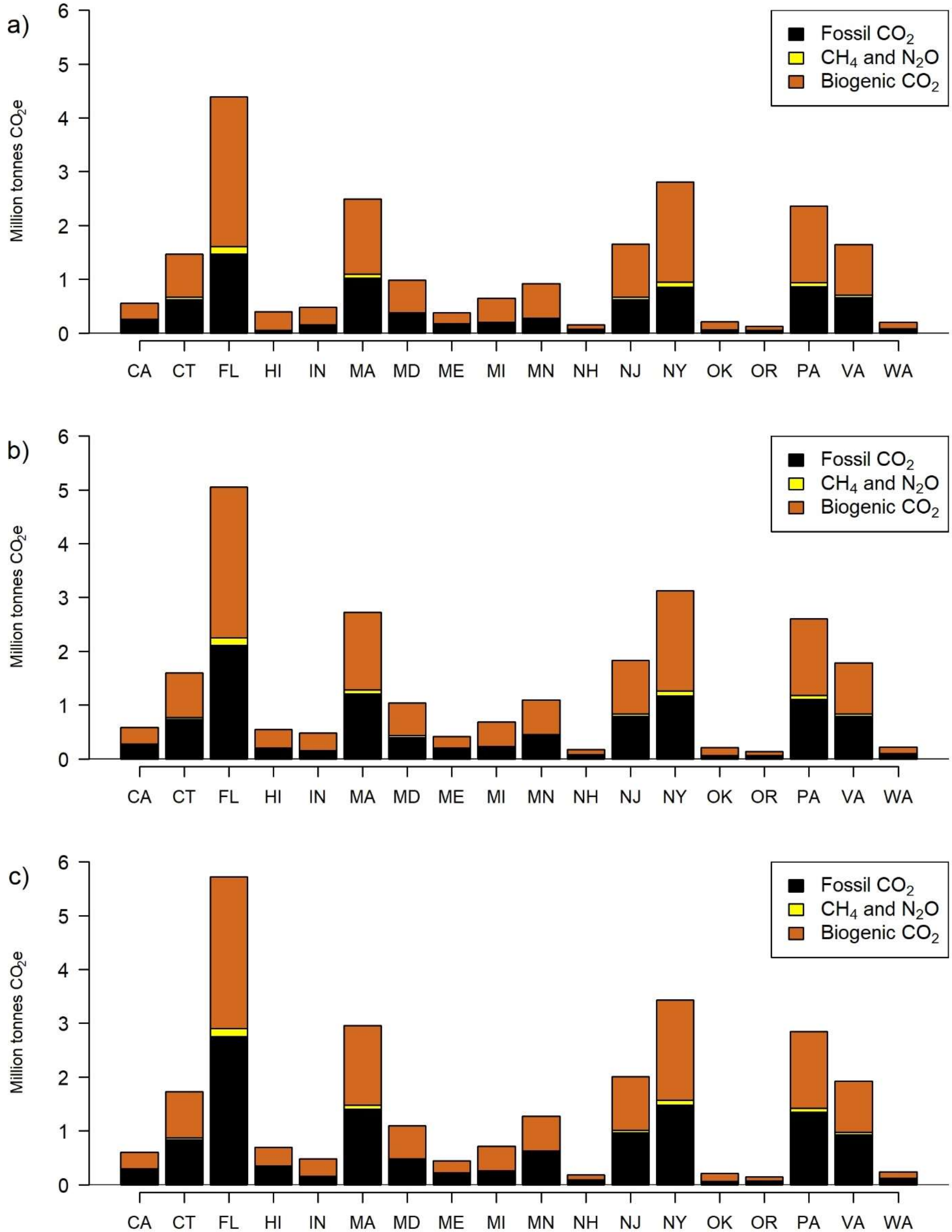

Figure 3: Excess GHG emissions from incinerators in 2018 by state for three electricity replacement scenarios: a) 100% replacement; b) 50% replacement; and c) 0% replacement. CHP incinerators are replaced at 100% of energy in all three scenarios.

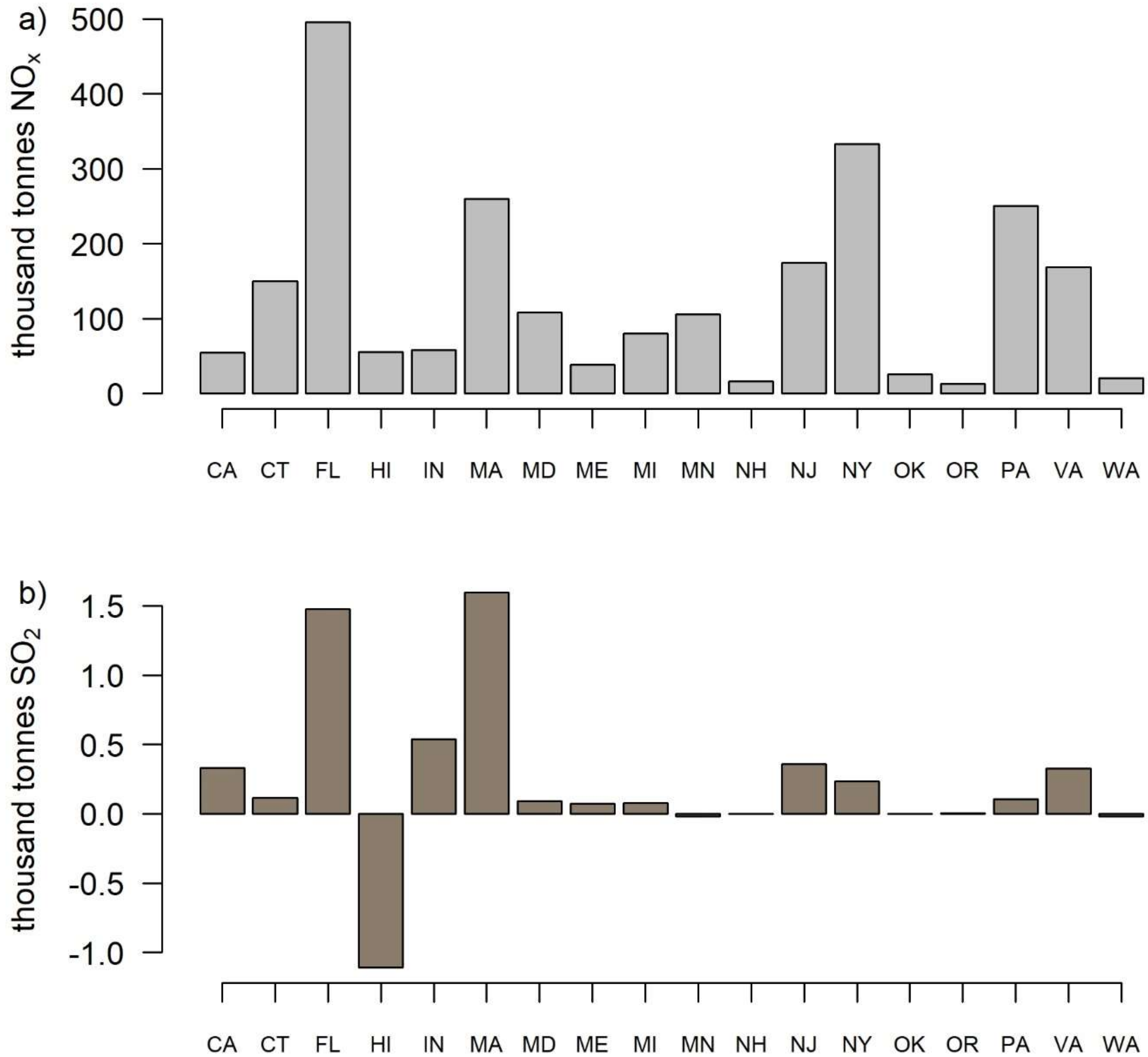

Figure 4: Excess NO_x (top) and SO_2 (bottom) emissions from incinerators in 2018 by state, under a 100% energy replacement scenario. Negative excess emissions indicates that incinerators emit less of the pollutant than replacement energy sources.

The average age of incinerators operating at the beginning of 2021 is 32 ± 5 years; 48 incinerators are over the age of 30. Only one incinerator – Palm Beach #2 – has begun operations since 1995; its estimated life extends through 2050. Retiring the current fleet of incinerators at end-of-life, estimated to be 35 years, will incur 157.1 million tonnes CO_2e, 16.8 million tonnes NO_x, and 39,700 tonnes SO_2 in excess emissions under the "no policy" scenario to 2050 (Figure 5, Table 2). In the Paris scenario, excess GHG emissions increase by 10.1 million tonnes, almost all fossil CO_2; and excess SO_2 emissions increase by 5900 tonnes. Extending each functioning incinerator's life by 20 years will incur 585.7 million tonnes CO_2e, 61.0 million tonnes NO_x and 125,500 tonnes SO_2 additional emissions under the "no policy" scenario and 637.7 million tonnes CO_2e, 61.9 million tonnes NO_x and 161,200 tonnes SO_2 under the Paris scenario.

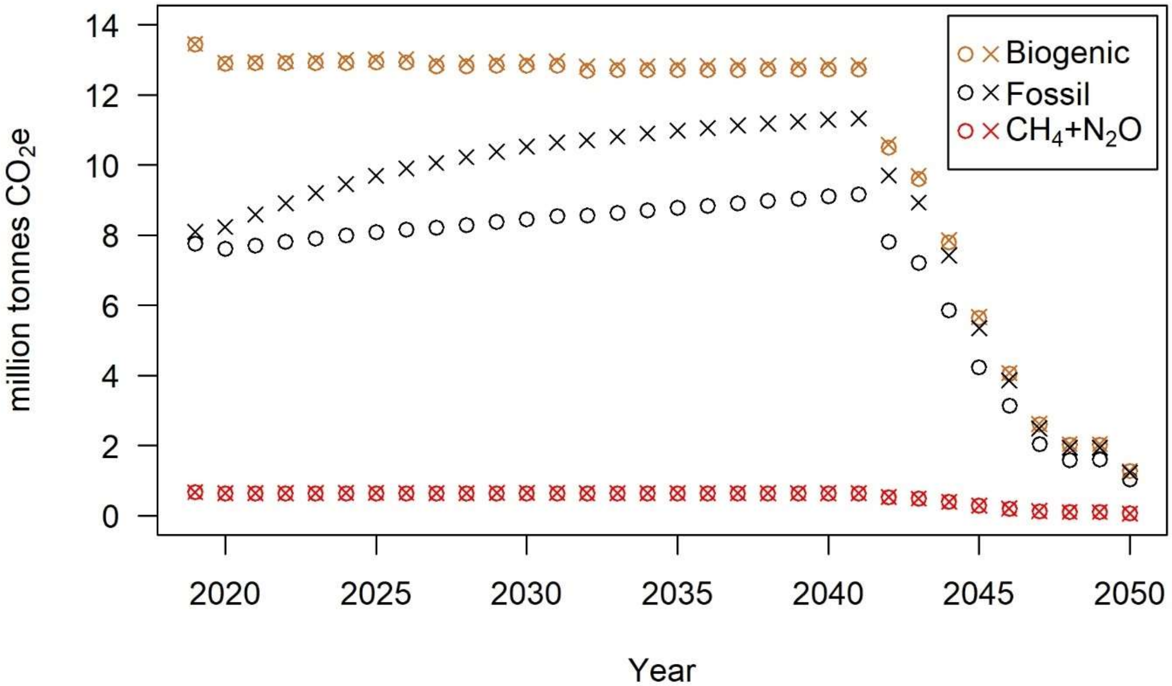

Figure 5: Annual excess GHG emissions resulting from extending incinerators' operating lives by 20 years. Open circles indicate the no policy scenario and crosses the Paris scenario, both with 100% energy replacement. The scenarios are almost identical for biogenic, N_2O, and CH_4 emissions.

Closure:	**End of Life**	**End of Life**	**Extended**	**Extended**
Decarbonization scenario:	**No policy**	**Paris**	**No policy**	**Paris**
Fossil CO_2	58.0	67.7	228.1	277.4
N_2O and CH_4	4.7	4.7	17.0	17.2
Biogenic CO_2	94.3	94.7	340.6	343.1
NO_x	16.8	17.0	61.0	61.9
SO_2	39.7	45.8	125.5	161.2
All GHGs	157.1	167.2	585.7	637.7

Table 2: Excess incinerator emissions under four scenarios to 2050. All data in million tonnes (CO_2e for N_2O, CH_4) except SO_2, which is in thousand tonnes. End of life is estimated at 35 years; "Extended" adds an additional 20 years.

4. Discussion

4.1 Incineration is a high-carbon power source

The intensity and excess emissions analyses indicate that incineration is the most emissions-intensive form of electricity production and its removal from the grid would improve overall grid performance. Even under conservative assumptions – that electricity-only incinerators are replaced by a mix of sources reflecting the 2018 average for each subregion, and CHP incinerators are replaced by natural gas – every category of emissions but SO_2 would be reduced in every subregion, and SO_2 would be reduced nationally. In fact, incinerators' contribution to the electric grid is minimal (0.3%) and could easily be replaced by renewable energy sources. This is particularly likely as incinerator removal would free up RPS subsidies to expand true renewables. As the grid decarbonizes, the benefits of incinerator shutdowns will further increase. Incineration cannot therefore be considered a "low-carbon" energy source, as it is currently designated in many state laws.

Excess NO_x and SO_2 emissions track excess GHG emissions well, with $r = 0.97$, $p \ll 0.001$ (NO_x with fossil CO_2) and $r = 0.73$, $p \ll 0.001$ (SO_2 with fossil CO_2) on a state-by-state basis. This indicates that incinerator removal would produce considerable co-benefits to host communities. Indeed, on this question, beneficial climate policy and beneficial environmental justice policy are indistinguishable.

4.2 Sensitivity Analysis

Consistent with earlier studies, the choice of replacement energy scenario made a dramatic difference in the excess emissions of fossil CO_2 and SO_2. However, it had little impact on other excess emissions. Compared with a 100% replacement scenario, the 0% replacement scenario increased excess fossil CO_2 emissions by 59%, CH_4 and N_2O by 3%, biogenic CO_2 by 2%, NO_x by 3%, and SO_2 by 80% (Table A2). This is due to the fact that, in 2018, replacement energy sources are major sources of fossil CO_2 and SO_2 but emit minimal levels of the other gases.

The primary factor driving excess emissions to 2050 is the lifespan of incinerators. Extending the incinerator fleet's operating life by 20 years increases total emissions by 3 to 4 times (average: 3.67) in each emissions category. By contrast, the choice of decarbonization scenario is significant but not large: in comparison with the "no policy" scenario, the faster, "Paris" pathway increases excess emissions by 6.5% in the end-of-life case and 8.9% in the extended life case. Indeed, as the 2018 analysis shows, incinerators produce excess emissions even without grid decarbonization. State-to-state variation in the magnitude of excess emissions is primarily due to variability in the emissions intensity of the replacement energy and whether the incinerator is a CHP facility. Nevertheless, every incinerator produces excess GHG emissions and therefore every state's GHG emissions intensity is higher than it would be without incinerators. These results are consistent with those of Tabata (2013), who analyzed 727 power-producing incinerators in Japan and found that only five did not produce excess fossil CO_2 emissions.

4.3 CHP plants

Our analysis raises questions about eGRID's reporting of thermal output from CHP plants. CHP plants are generally more energy-efficient than electricity-only plants and therefore show lower emissions intensity. For example, Pratt and Lenaghan (2020) report that Scotland's electricity-only incinerators are 61% more intensive than its heat-only incinerator; Healy (2012) concludes that CHP can deliver approximately 20% savings in emissions intensity for a grid intensity of ~0.45 kg CO_2/kWh; Kelly (2014) estimates 21% lower emissions intensity from CHP; while Jarre (2016) finds only marginal differences in emissions intensity between gas-fired CHP and electricity-only plants. However, our analysis shows that electricity-only plants are on average

3.5 to 4 times more emissions-intensive than CHP plants. The magnitude of this difference is highly anomalous.

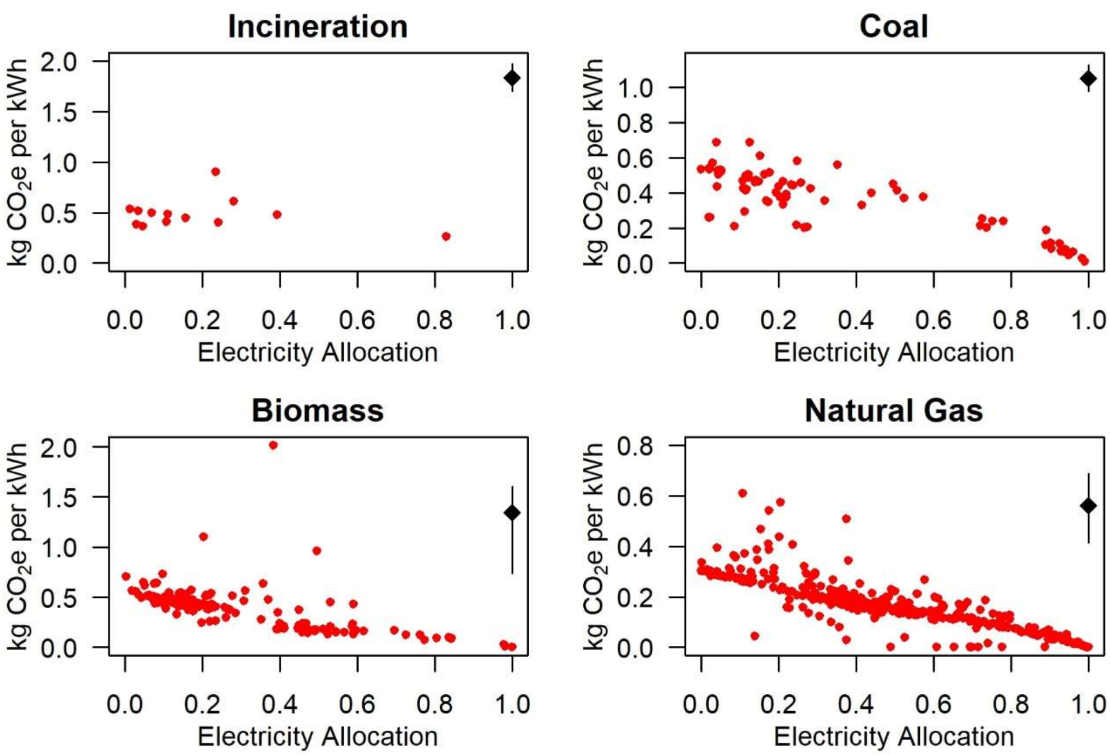

Figure 6: Relationship between energy allocation to electricity and emissions intensity for four fuel types. Each red dot indicates a CHP plant. Black diamonds and lines indicate the median value and interquartile range for electricity-only plants.

We would expect that CHP plants that dedicate more of their energy output to electricity generation would be less efficient and thus more emissions-intensive. Yet the data show the opposite: emissions intensity is negatively correlated with electricity allocation across fuel types (Figure 6). The discrepancy in emissions intensity between plants that devote at least 90% of their energy output to electricity and those that produce only electricity is particularly stark (Table 3). The reason for these inconsistencies is not clear, but may lie in the methodology for calculating plants' heat and/or steam output. In the eGRID database, the useful thermal output is not measured but estimated from fuel consumption and electricity production; uniform efficiency factors are applied to all facilities, obscuring significant real-world variations in efficiency (USEPA, 2020b). A systematic overestimation of the heat/steam exported would explain both discrepancies. It would also result in over-allocating emissions to heat/steam production, producing artificially low figures for electricity emissions intensity. To test this hypothesis and rectify any reporting bias, we suggest that the Energy Information Agency collect actual measurements of steam and heat output from CHP plants. Regardless of these discrepancies, incinerators still perform poorly compared to other cogeneration facilities.

Energy production	MSWI	Coal	Biomass	Natural Gas
100% electricity	1.92	1.10	1.28	0.74
90%+ electricity	n/a	0.06	0.01	0.02
All CHP plants	0.48	0.36	0.40	0.17

Table 3: Total GHG emissions intensity, in kg CO_2e/kWh, for plants that export only electricity (top row); CHP plants that use 90% or more of their energy for electricity production (middle row); and all CHP plants (bottom row). Total GHG emissions intensity is the sum of fossil CO_2, biogenic CO_2, CH_4, and N_2O emitted.

4.4 Biogenic emissions

Biogenic emissions are GHG emissions that result from the combustion or decomposition of non-fossilized biological material, such as wood, paper, cardboard, yard and food waste. In eGRID, CO_2 emissions from the combustion of landfill gas, which is primarily methane, are also classified as biogenic. 49% of incinerator CO_2 emissions are classified as biogenic in comparison to 2.8% for the national grid as a whole, reflecting the small role of biomass and biofuels in the nation's electricity system. The question of how, and even whether, to count biogenic emissions is controversial. The biomass industry argues that biogenic CO_2 emissions are inherently climate-neutral because they form part of a natural biological cycle; this position has found its way into policy, including the massive COVID relief bill passed in December 2020 (Cuellar, 2020). Such arguments ignore the overwhelming evidence that human perturbance of natural carbon cycles, including through deforestation and soil degradation, contribute significantly to atmospheric carbon loading (Ciais et al., 2014; Searchinger et al., 2009). As the radiative forcing of atmospheric CO_2 is virtually identical for biogenic and fossil CO_2, it is imperative to minimize emissions of both (United Nations Environment Programme, 2010). Accurate assessments of the climate impact of energy production thus require accounting for both biogenic and fossil CO_2. In its guidelines for national GHG emissions accounting, the Intergovernmental Panel on Climate Change requires reporting biogenic CO_2 emissions separately from fossil fuel emissions but not including them in the power sector total, as this would lead to double-counting; such emissions are already counted under Agriculture, Forestry, and Other Land Uses (Garg & Weitz, 2019). Here, we follow IPCC guidance and report biogenic emissions separately, since the goal of carbon neutrality necessarily requires zeroing out both fossil fuel and biogenic emissions.

4.5 Role of renewable energy subsidies

One of the aims of RPS programs is to diversify energy sources by incentivizing the construction of new renewable energy capacity (Upton & Snyder, 2017). In the case of incineration, this has failed: incinerator construction in the U.S. all but halted in the mid-1990s, just as state-level RPS laws became widespread. The issuance of RECs to incinerators is thus a form of economic rent: a subsidy to pre-existing facilities that produces no additional societal benefit.

REC sales data are fragmented and not public, so it is difficult to assess the financial impact of RPS programs (Mack et al., 2011). We estimate the value of REC sales to the incinerator industry at $41 to $44 million in 2018 (see Appendix for details). Removing incinerators from existing RPS programs would free up substantial subsidies for zero-emissions forms of power generation.

At an average age of 32 years, most of the U.S. incinerator fleet is nearing retirement age. 31 incinerators retired between 2000 and 2020, at an average age of 25 ± 6 years. In the next few years, most incinerator operators will have to decide whether to decommission their facilities or invest extensive capital in refurbishment to extend their lifespans. Waste industry analyses suggest that the availability of subsidies, in particular inclusion in state- or a national-level RPS program, would be the deciding factor in most decommissioning decisions (Baptista & Perovich, 2019; Karidis, 2016; McAnulty, 2019). To evaluate the impact of continuing to subsidize waste incineration, we calculated the excess emissions of each incinerator to the end of its expected operating life, which we took to be 35 years, and the additional excess emissions that would result from extending each incinerator's operations for another 20 years. The results indicate that extending the operating life of incinerators would be counterproductive to the aim of decarbonizing the electric grid. On the other hand, excluding incinerators from existing RPS programs might prompt many of them to close early. Even if replaced by fossil fuel plants, this would still result in reduced emissions.

4.6 Caveat

An important caveat in this analysis is that it examines only the direct emissions from power generation, excluding the emissions associated with the production and transport of the fuels. These "upstream" emissions can be on par with direct emissions, particularly in the case of natural gas (Alvarez et al., 2018). A full accounting of production-related emissions for municipal waste would be quite complex and outside the scope of this study.

5. Conclusions and Policy Implications

Incinerators are the most emissions-intensive form of generating electricity in the U.S. today. This is true regardless of the methodology employed (such as omitting biogenic emissions, using a different timescale for GWP, or analyzing subregions separately). As such, they are the last energy source that should be incentivized through renewable or clean energy policies. Incineration's inclusion in current, state-level RPS programs has not led to a build-out of incineration but may well be keeping alive the existing incinerator fleet; it has certainly diverted subsidies from non-combustion energy sources that would have lowered overall grid emissions. As these incinerators age, the availability of state and federal policies may be the deciding factor in whether or not to prolong their operational lives. To lower emissions, legislators should remove incineration from existing RPS and other subsidy programs, and avoid including them in any future federal subsidy program such as a clean energy standard. Incinerator closures would result in both a cleaner electric grid and less air pollution in environmental justice communities.

6. Acknowledgments

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7. References

Alvarez, R. A., Zavala-Araiza, D., Lyon, D. R., Allen, D. T., Barkley, Z. R., Brandt, A. R., et al. (2018). Assessment of methane emissions from the U.S. oil and gas supply chain. *Science*, *361*(6398), 186–188. https://doi.org/10.1126/science.aar7204

Astrup, T. F., Tonini, D., Turconi, R., & Boldrin, A. (2015). Life cycle assessment of thermal Waste-to-Energy technologies: Review and recommendations. *Waste Management*, *37*, 104–115. https://doi.org/10.1016/j.wasman.2014.06.011

Baptista, A. I., & Perovich, A. (2019). *U.S. Municipal Solid Waste Incinerators: An Industry in Decline*. The New School Tishman Environment and Design Center.

Barbose, G. (2019). *U.S. Renewables Portfolio Standards: 2019 Annual Status Update* (No. 48). Lawrence Berkeley National Laboratory.

Barbose, G., Wiser, R., Heeter, J., Mai, T., Bird, L., Bolinger, M., et al. (2016). A retrospective analysis of benefits and impacts of U.S. renewable portfolio standards. *Energy Policy*, *96*, 645–660. https://doi.org/10.1016/j.enpol.2016.06.035

Behrsin, I. (2019). Green Energy from Garbage? A Case Study of Municipal Solid Waste's Contested Inclusion in Maryland's Renewable Portfolio Standard. *Case Studies in the Environment*, *3*(1), 1–7. https://doi.org/10.1525/cse.2019.002048

Behrsin, I. (2020). Controversies of justice, scale, and siting: The uneven discourse of renewability in Austrian waste-to-energy development. *Energy Research & Social Science*, *59*, 101252. https://doi.org/10.1016/j.erss.2019.101252

Brown, M. (2014, December 5). Is waste a source of renewable energy? Retrieved January 26, 2021, from https://www.isonomia.co.uk/is-waste-a-source-of-renewable-energy/

Business as unusual. (2017). *Nature Energy*, *2*(9), 1–1. https://doi.org/10.1038/nenergy.2017.150

Ciais, P., Sabine, C. L., Bala, G., Bopp, L., Brovkin, V., Canadell, J., & Chhabra, A. (2014). Carbon and other biogeochemical cycles. In *Climate change 2013: the physical science basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 465–570). Cambridge University Press.

Covanta Holding Corporation Reports 2019 Fourth Quarter and Full Year Results And Provides 2020 Guidance. (2020, February 20). Retrieved January 26, 2021, from https://investors.covanta.com/2020-02-20-Covanta-Holding-Corporation-Reports-2019-Fourth-Quarter-and-Full-Year-Results-And-Provides-2020-Guidance

Cuellar, H. Consolidated Appropriations Act, Pub. L. No. H.R. 133 (2020). Retrieved from https://www.congress.gov/bill/116th-congress/house-bill/133

Database of State Incentives for Renewables & Efficiency. (n.d.). N.C. Clean Energy Technology Center. Retrieved from https://www.dsireusa.org/

Facility Level Information on GreenHouse gases Tool. (2020). U.S. Environmental Protection Agency. Retrieved from https://ghgdata.epa.gov/ghgp/main.do

Food and Water Watch. (2018). *Cleanwashing: How States Count Polluting Energy Sources as Renewable*.

Garg, A., & Weitz, M. M. (2019). *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Volume 2: Energy*. Intergovernmental Panel on Climate Change.

Grosso, M., Motta, A., & Rigamonti, L. (2010). Efficiency of energy recovery from waste incineration, in the light of the new Waste Framework Directive. *Waste Management*, *30*(7), 1238–1243. https://doi.org/10.1016/j.wasman.2010.02.036

Grubert, E. (2020). Fossil electricity retirement deadlines for a just transition. *Science*, *370*(6521), 1171–1173. https://doi.org/10.1126/science.abe0375

Healy, D. P. (2012). Influence of the carbon intensity of electricity on carbon savings from CHP. *Building Research & Information*, *40*(3), 317–326. https://doi.org/10.1080/09613218.2012.682418

Hochman, G., Rajagopal, D., & Zilberman, D. (2010). The Effect of Biofuels on Crude Oil Markets. *The Journal of Agrobiotechnology Management & Economics*, *13*(2). Retrieved from https://pubag.nal.usda.gov/catalog/286715

Jarre, M., Noussan, M., & Poggio, A. (2016). Operational analysis of natural gas combined cycle CHP plants: Energy performance and pollutant emissions. *Applied Thermal Engineering*, *100*, 304–314. https://doi.org/10.1016/j.applthermaleng.2016.02.040

Karidis, A. (2016, March 24). The 50 states of waste: How waste-to-energy definitions vary across the nation. *Waste Dive*. Retrieved from https://www.wastedive.com/news/the-50-states-of-waste-how-waste-to-energy-definitions-vary-across-the-nat/416197/

Kelly, K. A., McManus, M. C., & Hammond, G. P. (2014). An energy and carbon life cycle assessment of industrial CHP (combined heat and power) in the context of a low carbon UK. *Energy*, *77*, 812–821. https://doi.org/10.1016/j.energy.2014.09.051

Mack, J. H., Gianvecchio, N., Campopiano, M. T., & M. Logan, S. (2011). All RECs Are Local: How In-State Generation Requirements Adversely Affect Development of a Robust REC Market. *The Electricity Journal*, *24*(4), 8–25. https://doi.org/10.1016/j.tej.2011.04.007

McAnulty, H. (2019). A Dirty Waste—How Renewable Energy Policies Have Financed the Unsustainable Waste-to-Energy Industry. *Boston College Law Review*, *60*(1), 385.

Michaels, T. (2009). *Waste Not, Want Not: The Facts Behind Waste-to-Energy*. Energy Recovery Council.

Morris, J., Scott Matthews, H., & Morawski, C. (2013). Review and meta-analysis of 82 studies on end-of-life management methods for source separated organics. *Waste Management*, *33*(3), 545–551. https://doi.org/10.1016/j.wasman.2012.08.004

Open Data Platform. (n.d.). U.S. Energy Information Administration. Retrieved from https://www.eia.gov/opendata/

Powell, J. T., Townsend, T. G., & Zimmerman, J. B. (2016). Estimates of solid waste disposal rates and reduction targets for landfill gas emissions. *Nature Climate Change*, *6*(2), 162–165. https://doi.org/10.1038/nclimate2804

Pratt, K., & Lenaghan, M. (2020). *The climate change impacts of burning municipal waste in Scotland: Technical Report*. Zero Waste Scotland.

Ryan, N. A., Johnson, J. X., & Keoleian, G. A. (2016). Comparative Assessment of Models and Methods To Calculate Grid Electricity Emissions. *Environmental Science & Technology*, *50*(17), 8937–8953. https://doi.org/10.1021/acs.est.5b05216

Ryan, N. A., Johnson, J. X., Keoleian, G. A., & Lewis, G. M. (2018). Decision Support Algorithm for Evaluating Carbon Dioxide Emissions from Electricity Generation in the United States. *Journal of Industrial Ecology*, *22*(6), 1318–1330. https://doi.org/10.1111/jiec.12708

Schivley, G., Azevedo, I., & Samaras, C. (2018). Assessing the evolution of power sector carbon intensity in the United States. *Environmental Research Letters*, *13*(6), 064018. https://doi.org/10.1088/1748-9326/aabe9d

Schott, A. B. S., Wenzel, H., & Jansen, J. la C. (2016). Identification of decisive factors for greenhouse gas emissions in comparative life cycle assessments of food waste management – an analytical review. *Journal of Cleaner Production*, *119*, 13–24. https://doi.org/10.1016/j.jclepro.2016.01.079

Searchinger, T. D., Hamburg, S. P., Melillo, J., Chameides, W., Havlik, P., Kammen, D. M., et al. (2009). Fixing a Critical Climate Accounting Error. *Science*, *326*(5952), 527–528. https://doi.org/10.1126/science.1178797

Smith, A., Brown, K., Ogilvie, S., Rushton, K., & Bates, J. (2001). *Waste management options and climate change*. European Commission DG Environment.

Tabata, T. (2013). Waste-to-energy incineration plants as greenhouse gas reducers: A case study of seven Japanese metropolises. *Waste Management & Research*, *31*(11), 1110–1117. https://doi.org/10.1177/0734242X13502385

The Biden Plan to Build a Modern, Sustainable Infrastructure and an Equitable Clean Energy Future. (n.d.). Retrieved January 22, 2021, from https://joebiden.com/clean-energy/

United Nations Environment Programme. (2010). *Waste and Climate Change: Global Trends and Strategy Framework*. Osaka/Shiga.

Upton, G. B., & Snyder, B. F. (2017). Funding renewable energy: An analysis of renewable portfolio standards. *Energy Economics*, *66*, 205–216. https://doi.org/10.1016/j.eneco.2017.06.003

U.S. Energy Information Administration. (2020). *Annual Energy Outlook 2020*. Retrieved from https://www.eia.gov/outlooks/aeo/pdf/aeo2020.pdf

U.S. House Select Committee on the Climate Crisis. (2020). *Solving the Climate Crisis: The Congressional Action Plan for a Clean Energy Economy and a Healthy and Just America*. Retrieved from https://climatecrisis.house.gov/report

USEPA. (2006). *Solid Waste Management and Greenhouse Gases: a Life-Cycle Assessment of Emissions and Sinks* (p. 170). US Environmental Protection Agency.

USEPA. (2009). *Opportunities to Reduce Greenhouse Gas Emissions through Materials and Land Management Practices* (p. 98). US Environmental Protection Agency.

USEPA. (2020a). Emissions & Generation Resource Integrated Database (eGRID) (Version 2018 v2). Retrieved from https://www.epa.gov/egrid

USEPA. (2020b). *The Emissions & Generation Resource Integrated Database: Technical Support Document for eGRID with Year 2018 Data*.

York, R. (2012). Do alternative energy sources displace fossil fuels? *Nature Climate Change*, *2*(6), 441–443. https://doi.org/10.1038/nclimate1451

APPENDIX

Text: Methodological Notes

We used annual emissions and electricity generation data from the U.S. Environmental Protection Agency's 2018 Emissions and Generation Resource Integrated Database version 2 (eGRID), aggregated at the plant level. We excluded plants, including four incinerators, that had zero or negative net electricity generation (Table A3). We identified a further nine incinerators that are not included in the eGRID database; of these, we were able to find sufficient data to include three in our analysis (Table A4). We used unadjusted, plant-level data, deducting biogenic carbon dioxide (CO_2) emissions from total CO_2 emissions to calculate fossil fuel CO_2 emissions. We excluded 17 facilities, none of them incinerators, whose emissions data showed internal inconsistencies; this made no significant difference to the results. To express methane (CH_4) and nitrous oxide (N_2O) emissions in CO_2e, we used the most recent global warming potentials (GWP) based on the conventional 100-year timeline, including climate-carbon feedbacks: 34 for CH_4 and 298 for N_2O (IPCC AR5 WG1 page 714).

In calculating emissions past 2018, we did not extend the operating lifespans of the four incinerators that shut down in 2018 and 2019: Commerce, Detroit, Elk River, and Warren. The Harrisburg and Tulsa incinerators had already undergone major retrofits in 2006 and 2011 respectively, and we took their expected and extended lifespans both to be 20 years from the dates of those retrofits.

We estimated the value of REC sales two ways, assuming that all electricity generated in incinerators translated into REC sales. One, we extrapolated Covanta's publicly-reported $12 million in REC sales to the entire industry on the basis of MWh generated ("Covanta Holding Corporation Reports 2019 Fourth Quarter and Full Year Results And Provides 2020 Guidance," 2020). Two, we calculated an average REC price of $3.10 from the states with publicly available data: Tier 1 from Maryland and Tier 2 from Connecticut, Maine, Massachusetts, Pennsylvania, and New Jersey (Barbose, 2019). These methods resulted in estimates of $41 million and $44 million respectively.

State	Includes incinerators?	Notes
Arizona	Yes	Language unclear, but allows oxidation and gasification of municipal waste
California	No	RECs for Stanislaus County incinerator generated prior to 2017 are grandfathered. RECs for non-combustion conversion of solid waste are allowed.
Colorado	Yes	For pyrolysis of MSW
Connecticut	Yes	
Delaware	No	
Guam	Yes	
Hawaii	Yes	MSW legally defined as biomass
Illinois	No	
Indiana	Yes	
Iowa	Yes	

Kansas	No	
Maine	Yes	
Maryland	Yes	The only state that awards Tier 1 RECs to incinerators
Massachusetts	Yes	
Michigan	Yes	Only for gasification and plasma gasification facilities
Minnesota	Yes	
Missouri	Yes	"Thermal depolymerization and pyrolysis" of MSW are eligible
Montana	No	Biomass plants that burn chemically-treated wood are eligible
Nevada	Yes	Municipal solid waste legally defined as biomass
New Hampshire	Yes	Pyrolysis and gasification eligible but not incineration
New Jersey	Yes	
New Mexico	No	
New York	No	
North Carolina	No	
North Dakota	No	
Northern Marianas Islands	Yes	Municipal solid waste legally defined as biomass
Ohio	No	Energy from solid waste "through fractionation, biological decomposition, or other process that does not principally involve combustion" is eligible.
Oklahoma	Yes	Municipal solid waste is legally defined as biomass
Oregon	Yes	Only for incinerators established before 1995
Pennsylvania	Yes	
Puerto Rico	Yes	
Rhode Island	No	
South Carolina	?	Not explicitly included or excluded; biomass included but undefined
South Dakota	Yes	
Texas	No	
US Virgin Islands	Yes	Language unclear
Utah	Yes	
Vermont	No	
Virginia	Yes	
Washington	No	
Washington, DC	No	
Wisconsin	Yes	Plasma gasification and refuse-derived fuel are eligible

Table A1: State-level jurisdictions with RPS programs. Sources: State statutes; Database of State Incentives for Renewables & Efficiency (DSIRE, n.d.); Food and Water Watch (2018).

Gas	100%	50%	0%	Change
Fossil CO_2	7.7	10.0	12.3	59%
$CH_4 + N_2O$	0.7	0.7	0.7	3%
Biogenic CO_2	13.5	13.6	13.7	2%
NO_X	2.4	2.5	2.5	3%
SO_2	4.2	5.9	7.5	80%
GHGs	21.9	24.3	26.7	22%

Table A2: 2018 Excess emissions from incinerators for three energy replacement scenarios: 100% replacement, 50% replacement, and no replacement. "Change" indicates the change in excess emissions between the no replacement and 100% replacement scenarios. GHGs is the total of all greenhouse gases. Data in million tonnes (CO_2e for CH_4, N_2O) except SO_2 which is in thousand tonnes.

Facility name	City	State	In eGRID?	Reason for exclusion
Covanta Huntsville/Huntsville Solid Waste-to-Energy Facility	Huntsville	AL	No	Electricity generation not available
Wheelabrator North Broward	Coconut Creek	FL	Yes	Facility closed
Arnold O. Chantland Resource Recovery Plant	Ames	IA	No	Emissions data not available
Perham Incinerator	Perham	MN	Yes	No net electricity generation
Polk County Solid Waste Resource Recovery Facility	Fosston	MN	No	Emissions data not available
Wheelabrator Claremont Facility	Claremont	NH	Yes	Facility closed
Refuse & Coal	Columbus	OH	Yes	Facility closed
Hampton-NASA Steam Plant	Hampton	VA	No	Steam production data not available
Barron County WTE & Recycling Facility	Almena	WI	No	Emissions data not available
Xcel Energy French Island Generating Plant	LaCrosse	WI	Yes	Fuel mix not available

Table A3: Incinerators excluded from the analysis.

Facility name	City	State	Fossil CO_2	Bio-genic CO_2	CH_4	N_2O	Net gen-eration (MWh)	Heat pro-duction (mmBTU)
Covanta Niagara	Niagara Falls	NY	303,568	405,471	248	32	161,508	6,105,771
Pittsfield Resource Recovery Facility	Pittsfield	MA	25,251	39,490	23	3	4,380	637,474
Pope/ Douglas SWM	Alexandria	MN	27,289	31,222	20	3	7,884	713,747

Table A4: Non-eGRID incinerators added to the analysis. Emissions data from EPA's Facility Level Information on GreenHouse gases Tool (FLIGHT; *Facility Level Information on GreenHouse gases Tool*, 2020). Steam production for Pittsfield and Pope/Douglas from FLIGHT. Niagara electricity generation from the Energy Information Agency's Open Data platform (*Open Data Platform*, n.d.). Remaining data calculated with EIA formulae and nameplate capacity.

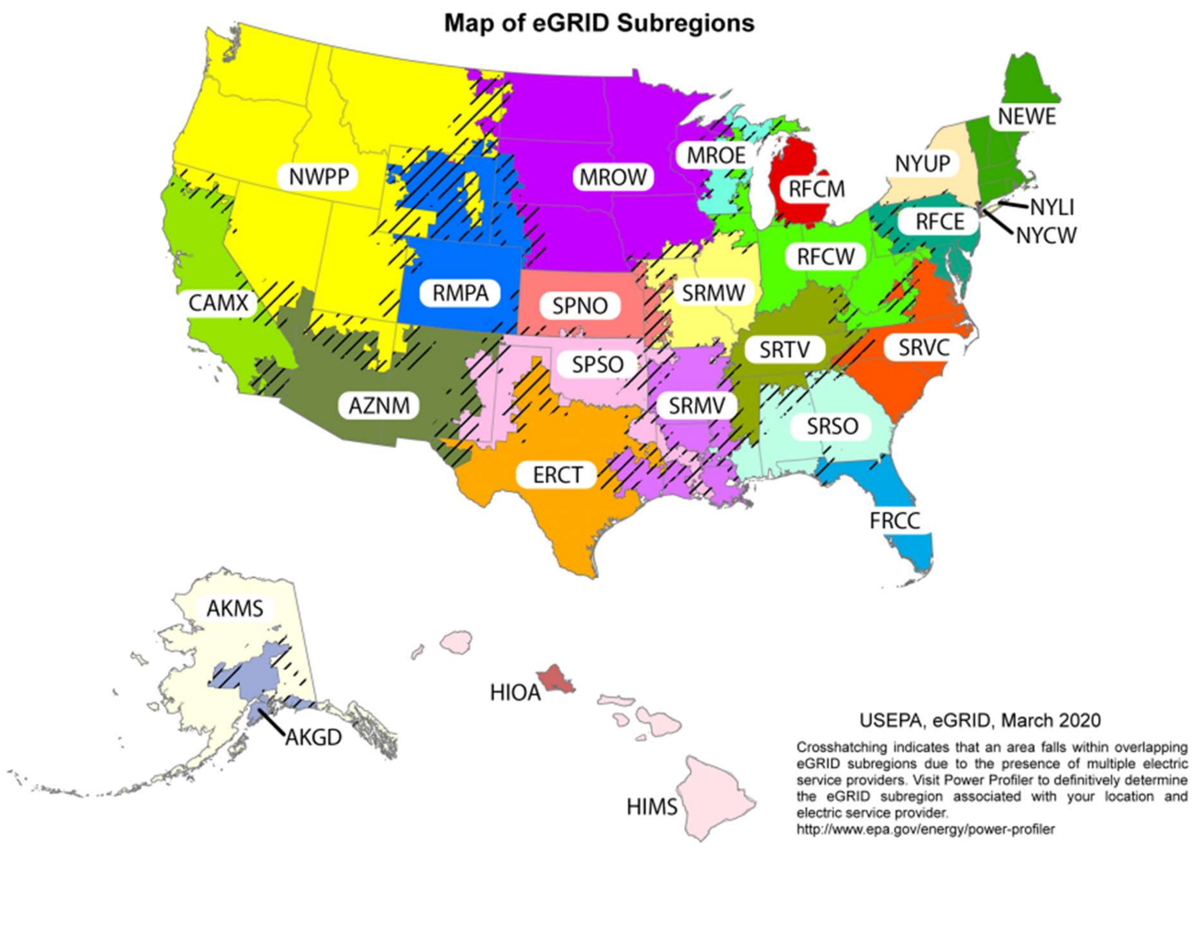


Figure A1: Map of eGRID subregions used in this analysis. Source: USEPA.

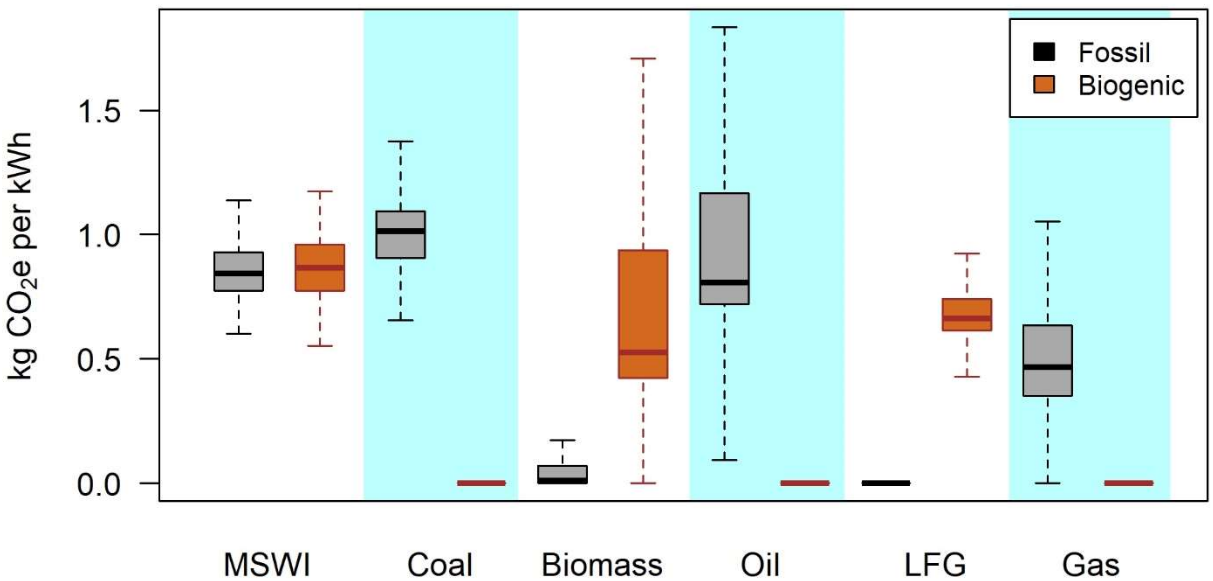
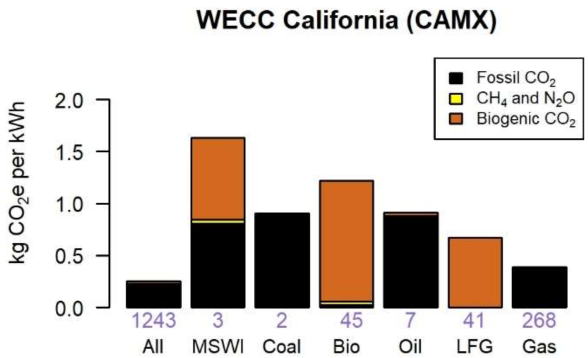
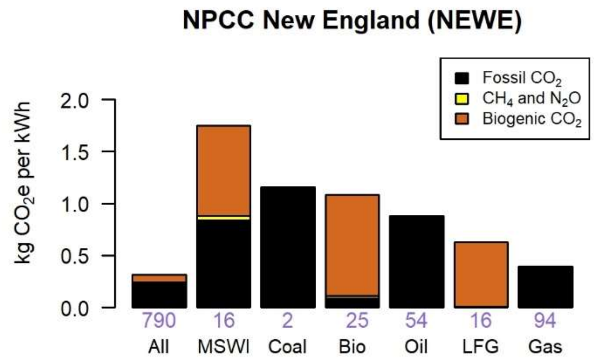
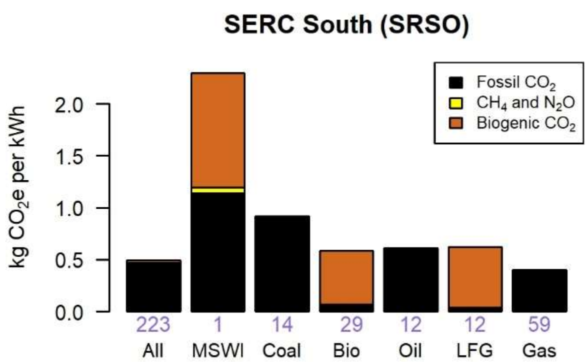


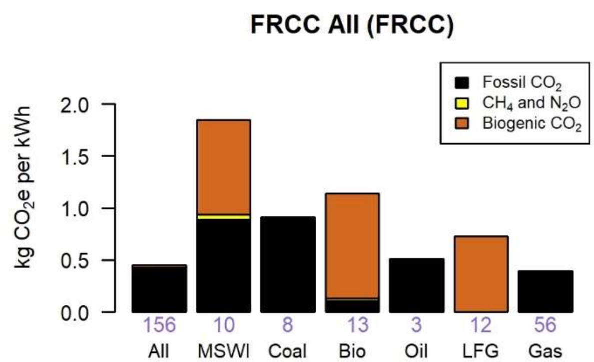
Figure A2: GHG emissions intensity at the plant level (cf. generation-weighted data in Figure 1). CH_4 and N_2O emissions are not depicted. MSWI is incineration, LFG is landfill gas, and Gas is natural gas. Heavy line indicates the median value and boxes the interquartile range, and whiskers the outliers.

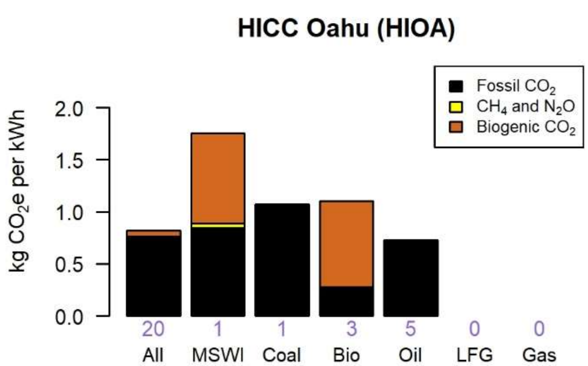
Figure A3 (following pages): Generation-weighted mean subregional GHG emissions intensity by major fuel type for electricity. "MSWI" is municipal solid waste incineration, "LFG" is landfill gas, and "Gas" refers to natural gas. Purple numbers indicate the number of plants in the named subregion. See map (Figure A1) for subregions.

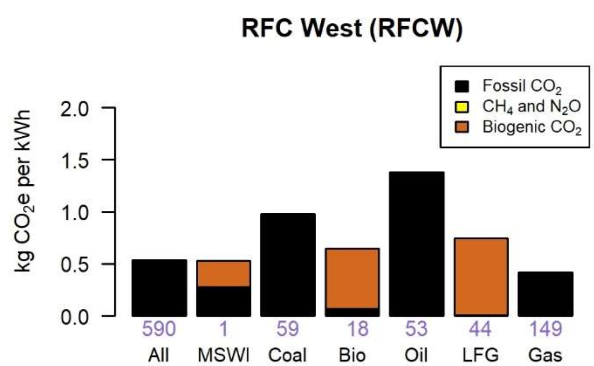

WECC California (CAMX)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
1243 3 2 45 7 41 268
All MSWI Coal Bio Oil LFG Gas

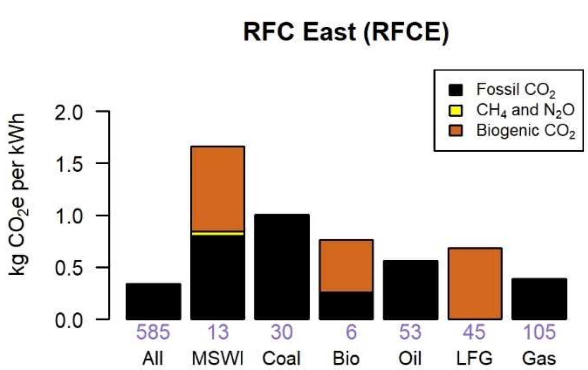

NPCC New England (NEWE)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
790 16 2 25 54 16 94
All MSWI Coal Bio Oil LFG Gas

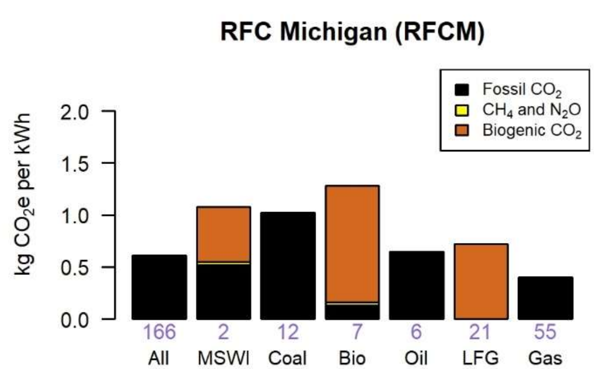

SERC South (SRSO)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
223 1 14 29 12 12 59
All MSWI Coal Bio Oil LFG Gas

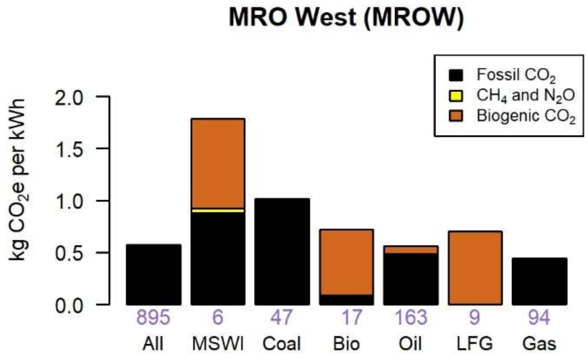

FRCC All (FRCC)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
156 10 8 13 3 12 56
All MSWI Coal Bio Oil LFG Gas

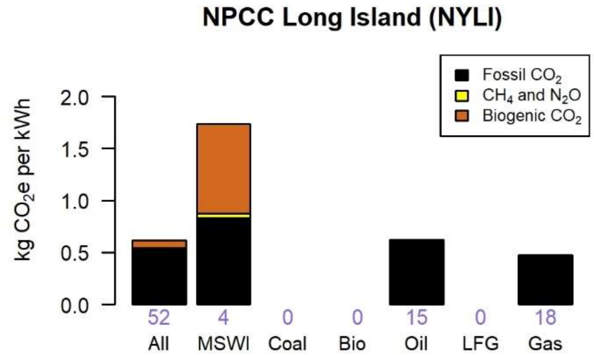

HICC Oahu (HIOA)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
20 1 1 3 5 0 0
All MSWI Coal Bio Oil LFG Gas

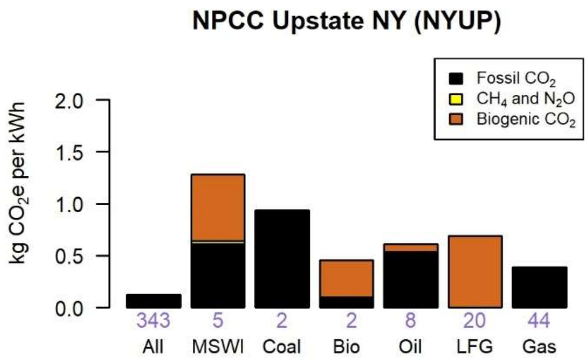

RFC West (RFCW)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
590 1 59 18 53 44 149
All MSWI Coal Bio Oil LFG Gas

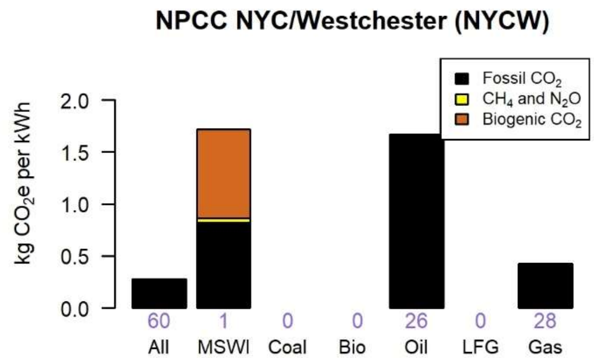

RFC East (RFCE)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
585 13 30 6 53 45 105
All MSWI Coal Bio Oil LFG Gas

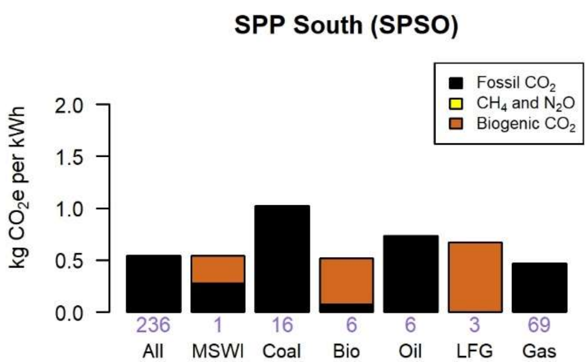

RFC Michigan (RFCM)
Fossil CO2
CH4 and N2O
Biogenic CO2
kg CO2e per kWh
0.0
0.5
1.0
1.5
2.0
166 2 12 7 6 21 55
All MSWI Coal Bio Oil LFG Gas

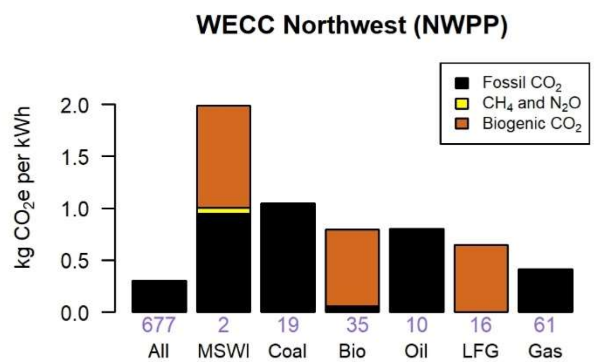

MRO West (MROW)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
895 6 47 17 163 9 94
All MSWI Coal Bio Oil LFG Gas

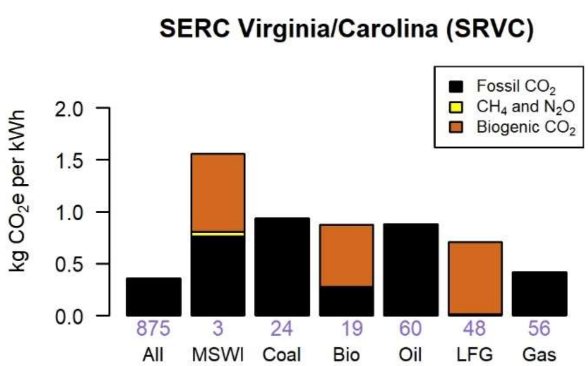

NPCC Long Island (NYLI)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
52 4 0 0 15 0 18
All MSWI Coal Bio Oil LFG Gas


NPCC Upstate NY (NYUP)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
343 5 2 2 8 20 44
All MSWI Coal Bio Oil LFG Gas


NPCC NYC/Westchester (NYCW)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
60 1 0 0 26 0 28
All MSWI Coal Bio Oil LFG Gas


SPP South (SPSO)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
236 1 16 6 6 3 69
All MSWI Coal Bio Oil LFG Gas


WECC Northwest (NWPP)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
677 2 19 35 10 16 61
All MSWI Coal Bio Oil LFG Gas


SERC Virginia/Carolina (SRVC)
Fossil CO_2
CH_4 and N_2O
Biogenic CO_2
kg CO_2e per kWh
2.0 1.5 1.0 0.5 0.0
875 3 24 19 60 48 56
All MSWI Coal Bio Oil LFG Gas