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Technology Diffusion in Carbon Markets: Evidence from Aviation

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Abstract
Carbon pricing has been found mainly to foster low-carbon innovation but not low-carbon technology adoption. Focusing on the aviation sector, a hard-to-abate industry, we provide novel evidence that the EU's Emission Trading System (EU ETS) is responsible for a greater diffusion of available low-carbon technologies. By exploiting a policy change in the carbon market scope, we find a more intensive use of efficient aircraft and a sizeable effect of aircraft retrofitting – use of winglets – compared to the counterfactual, driving improvements in emission intensity. These effects, however, are not uniform: aircraft choice and retrofitting decisions differ between Eastern and Western routes. Examining the mechanisms behind our findings, we show that the more intensive use of efficient aircraft—either through fleet replacement or differential use—is the main factor driving the ETS's effect on emission intensity. Our study highlights the importance of carbon pricing to accelerate the adoption rates of low carbon technologies, even in hard-to-abate sectors.

Keywords Carbon pricing · EU ETS · Aviation · Technology diffusion, diff-in-diffs

JEL Codes Q54 · D22 · L93

1 Introduction

In a world coping with an often-insufficient political consensus to keep global warming below 2 °C (IPCC 2022), the pivotal role of low-carbon technologies becomes even more critical in efforts to tie mitigation pledges to climate targets. In theory, when faced with a carbon price, firms avail themselves of abatement options for which costs are below the

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carbon price. This incentivizes polluting agents to pick the most cost-effective options to reduce their emissions. Alternatively, firms may seek to accommodate their output to their new marginal costs to continue to maximize profits. This search for cost-effectiveness is expected to encourage low-carbon innovation that should further reduce the costs of compliance in the long run. However, unlocking the full impact of low-carbon technologies requires not just innovation, but widespread adoption across firms. This paper examines technology adoption in response to carbon pricing by the EU's Emission Trading System (EU ETS).

The technological response to carbon pricing may take the form of inventing and innovating new low-carbon technologies or, alternatively, adopting existing low-carbon technologies, i.e., the diffusion of previous innovation processes. Notably, the EU's Emission Trading System (EU ETS) has been reported as generating more innovation than adoption (Calel 2020; Teixidó et al. 2019). Firms faced with having to pay a carbon price have been found to invest more heavily in R&D and to register more low-carbon patents than they would have done had the carbon price not been in place (Calel 2020; Calel and Dechezleprêtre 2016). Effects on adoption, however, are less conclusive, with mixed and country-specific results, mostly centered on the manufacturing sector.

Because data on firms' technological choices is rarely available, effects on adoption are mainly proxied by changes in emission intensities. However, these changes do not necessarily involve technological change. This is the case, for instance, in the electricity sector, in which the EU ETS is found to improve emission intensity thanks to fuel switching rather than actual technological change (Cao et al. 2021; Berghmans et al. 2014; Delarue et al. 2008; Ellerman and McGuinness 2008). In the manufacturing sector, evidence is mixed: the carbon market induced changes in emission intensities in Germany, France and, very slightly, in Lithuania (Petrick and Wagner 2014; Colmer et al. 2024; Jaraite and Di Maria 2016), but not in the UK or Norway (Calel 2020; Klemetsen et al. 2020). Yet, when found, changes in emissions intensity are often not driven by actual technological upgrade but by behavioral and operational changes (Petrick and Wagner 2014; Lofgren et al. 2014). Only a few studies couple the improvement in emission intensity with concurrent changes in tangible capital assets, providing suggestive, but not conclusive, evidence of low-carbon technology adoption (Colmer et al. 2024). Importantly, emission intensity is measured in all these studies in terms of emissions per employment level or per value added, which is not free of drawbacks. Metrics based on emissions per unit of physical output are less common and harder to compare, especially across heterogeneous manufacturing industries (which have been the focus of such studies).

In this paper, we investigate the effect of the EU ETS on actual technology diffusion using reliable, comparable metrics of technology change across Europe. We focus on the commercial aviation sector and analyze how the EU ETS affects realized emission intensity (measured in kg of CO_2 per seat-km), and how these changes are mediated by aircraft model choices and retrofit decisions, such as winglet adoption.

Aviation is not only a relevant case in its own right[1] but also offers useful insights into some of the broader challenges of low-carbon technology diffusion. Like other hard-to-

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[1] Aviation is the most climate-intensive mode of transport and, while today it accounts for only 6% of global climate impact (Lee et al. 2021), its projected growth −2050 flights are projected to be between 20% and 76% higher than 2019 levels– jointly with the environmental improvement in other sectors, means its global GHG contribution is set to grow critically in the coming years (IATA 2022; ICCT 2022).

abate sectors—such as maritime shipping, long-distance freight, and heavy industry—aviation is characterized by long-lived capital stock, international regulatory exposure, and limited short-run abatement opportunities. While aviation also has unique institutional and technological features, the parallels suggest it can serve as a valuable context for exploring how carbon pricing interacts with adoption incentives in capital-intensive, internationally integrated industries. Moreover, aviation offers several empirical advantages. Unlike other sectors, airlines operate largely homogenous fleets, and technology choices are more observable and comparable across operators and countries. Our analysis is not restricted to a single national context, as we draw on the universe of flights in Europe and neighboring countries, tracking aircraft, routes, and carriers over time.

To identify the impact of the EU ETS on aviation technology adoption, we exploit the policy change introduced when limiting the scope of the scheme for aviation, known as the 'stop-the-clock' law. Originally (that is, from January 1, 2012), any flight landing or taking off from an airport in the European Economic Area (EEA) was subject to EU ETS compliance and, hence, faced a carbon price. However, this generated considerable resistance among international carriers who deemed the regulation unlawful and a breach of the sovereignty of non-EU countries. In the face of significant international pressure, in April 2013, the European Commission proposed 'stopping the clock' on the ETS regulation for the aviation sector and limiting the scope of the EU ETS directive, retrospectively, to flights *within* the EEA, regardless of airline's nationality. This policy change, therefore, provides us with a group of flights that can be used as a control for our analysis.

We use a difference-in-differences strategy to compare changes in emissions intensity and changes in terms of technological and retrofitting available options in the aircraft fleets operating on the EU ETS regulated routes with those made by our control group between 2010 and 2021. Importantly, this identification strategy benefits from the fact that aircraft technology is rather homogenous across airlines and countries; above 70% of aircraft operated in Europe are manufactured by only two companies (Airbus and Boeing). As opposed to other sectors, this enhances comparability between treated and control groups.

Our results show that the EU ETS reduced average emission intensity (kg of CO_2 per seat-km) by 2–4% per year. This is a notable effect given the low carbon prices prevailing for most of the period, the sector's pre-existing downward trend in emission intensity, and the limited set of available technological options. We show this change is partly driven by changes in aircraft choices and retrofitting decisions.

The higher size effect was caused by the retrofitting of aircraft. We find an average increase of 10% in the share of aircraft that installed winglets. Winglets (the curved tips at the end of the wings) are the main retrofit in the air industry and improve a plane's aerodynamics by reducing drag and, consequently, fuel consumption.[2] In 2010, before the introduction of carbon pricing, winglets were fitted on about 10% of planes in Europe but, by 2021, this share had increased to above 30%. Here, we show that the EU ETS is responsible for a large share of this notable and rapid increase in the adoption of winglets.

Regarding the adoption of more efficient aircraft models, the EU ETS has increased the use of such models by 5% on average. These models have below-average carbon intensity for the year in question –thus accounting for the release of new aircraft models in the sam-

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[2] The concept was first devised in 1897, although the first Boeing and Airbus winglet models would not be employed until 1985. Earlier designs of winglets were not successful because the better aerodynamics was offset by the greater structural weight of the aircraft.

pled period– and over specific distances –thus accounting for air route length (for instance, for a given distance, models like A320 or B737-800 could be considered efficient in 2010 but not in 2021, when A320neo or B737-Max would have been regarded as efficient).

These effects are distributed differently across countries. Heterogeneity analyses show the EU ETS effect on emission intensity is mainly driven by Eastern European economies. When we split the treated group sample into flights within Western (Eastern) countries and compare them to similar flights in the control group, we find the carbon market reduced emission intensity by 10% in Eastern economies, while the effect is not statistically different from zero for flights within Western economies. Regarding the effects on aircraft choice and retrofitting decisions, we find the EU ETS increased the use of efficient aircraft on Eastern routes but not on Western routes. In contrast, winglet adoption was only significant in Western routes. One potential explanation for this differential effect is that airlines operating in Eastern Europe were further from the technological frontier, providing more opportunity to improve emission intensities.

Our results contribute to the broader literature examining the impact of carbon pricing on technology change. Here, we identify a causal link between carbon pricing and the diffusion of low-carbon technologies, an effect that has traditionally been more linked to environmental regulations (such as technology standards), but not to pricing instruments (Rausch and Yonezawa 2023; Clarke et al. 2006; del Río González 2009; Kerr and Newell 2003; Popp 2010; Snyder et al. 2003). Indeed, technology standards can provide fast, dependable outcomes; however, they lack the continuous incentives and cost-effectiveness that pricing instruments can offer. Technology diffusion is, after all, a slow and gradual process (Schumpeter 1942) but we show that carbon pricing can accelerate it, even in sectors with structural rigidities.

Fageda and Teixidó (2022) provide evidence of the short-run effect of the EU ETS on total emissions, reporting a 5% reduction compared to the counterfactual. They show that this short-run effect is primarily driven by the decrease in air supply,[3] with little scope for any abatement via technological change. In this paper, we shift the focus to the long-term response, characterized by retrofitting actions and technology upgrades, and the effects on emissions intensity. The extended post-treatment period (spanning nine years instead of four) and the detailed data on aircraft technology used enables us to analyze how carbon pricing interacts with the diffusion of low-carbon technologies.

As far as we know, only the study of De Jong (2022) has analyzed the role of the EU ETS on the adoption of low-carbon technologies in the aviation sector. Using airline-level data, he compares fleets ages between selected EU and US network carriers and finds that the EU ETS fosters the early replacement of older (narrowbody) aircraft.[4] In contrast, we use more granular data at the airline-aircraft-route level, which allows us to examine changes in emissions intensity and the specific deployment of aircraft across the entire European market.

Finally, our findings have important policy implications. Extensive evidence shows that air traffic volumes are closely tied to economic growth and wider economic benefits (e.g.; Brueckner 2003; Blonigen and Cristea 2015; Florida et al. 2015; Campante and Yanagizawa-Drott 2018). As a result, reducing emissions by limiting flights may result in signifi-

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[3] This result on air supply response to the EU ETS is further confirmed by Kang et al. (2022) using a synthetic control method.

[4] Some previous studies also provide evidence of airline fuel conservation actions in response to higher fuel prices (Brueckner and Abreu 2017, 2020; Brueckner et al. 2024; Fukui and Miyoshi 2017).

cant economic cost—making emissions reductions through technological improvements, a more attractive and less disruptive alternative.

The rest of this paper is organized as follows: Sect. 2 reviews the literature and basic theory on the economics of induced technology change related to both the EU ETS and to aviation; Sect. 3 presents our data and empirical strategy; Sect. 4 explains the identification strategy; Sect. 5 reports our main results; Sect. 6 analyzes the mechanisms that may drive our results; and Sect. 7 concludes and discusses policy implications.

2 EU ETS-Induced Technology Diffusion

2.1 Background

Technology change was described by Joseph Schumpeter (1942) as the result of three independent steps: *invention*, that is, the original development of a new technical idea; *innovation* or the penetration of the new technology into the market, thereby generating profits and/or a monopolistic position, and *diffusion*, that is, when the new technology is widely adopted by other firms in the market. Thus, the key difference between innovation and diffusion is that, while the former can be seen as a shift of the technological frontier, diffusion occurs as firms move towards that technological frontier (Jaffe et al. 2005). Here, we are specifically concerned with the latter. The EU ETS has been reported as playing a role in both invention and innovation, consistent therefore with the induced-innovation hypothesis, i.e. the change in the relative price of a factor spurs innovation aimed at economizing the use of said factor (Hicks 1932). But, here, our interest lies in testing the induced-diffusion hypothesis (Popp 2010).

While innovation can be “disruptive” (Schumpeter 1942), the diffusion of a new technology tends to be a slower process. Typically, adoption of a new technology follows an S-curve: an initial stage characterized by a handful of early adopters is followed by a period of mass-adoption, but when the market reaches maturity, the final phase returns to the earlier slow adoption rate (Geroski 2000). In the case of environmental technologies, empirical research indicates that the adoption of cleaner technologies only occurs when there is a need to comply with environmental policies, what the literature refers to as the induced-diffusion hypothesis (Kerr and Newell 2003; Popp 2010). What is not clear, however, is which instruments are best suited to achieve this goal (Popp et al. 2010; Lilliestam et al. 2021, 2022). Critically, while carbon prices have a relatively short history, innovation effects take a long time (van den Bergh and Savin 2021).

Calel and Dechezleprêtre (2016) report that the EU ETS increased innovation –measured in terms of the number of low-carbon patents– in the manufacturing sector. EU ETS firms generated 10% more patents of this nature than in the counterfactual. Similarly, Calel (2020) reports a 25% increase in low-carbon patents in the UK manufacturing sector and a similar impact on R&D spending. However, the same author fails to find any significant effect on emission intensity, which is relevant as this would point to the adoption of new technologies. The absence of any impact on emission intensity is further confirmed for Norway (Klemetsen et al. 2020), Sweden (Löfgren et al. 2014) and Lithuania (Jaraite and di Maria 2016). In Germany (Petrick and Wagner 2014) and France (Colmer et al. 2024), the EU ETS has been found to reduce emission intensities in the manufacturing sector, as measured in terms of value added, by 18% and 17% respectively. For Germany, fuel switching and other energy optimization processes seem to drive the improvement in emission intensity. For France, the reduction in emission intensity comes with a positive effect on capital stock which, though not conclusive as evidence, is consistent with firms adopting new production technology. Yet none of the authors mentioned can directly identify the effects of the policy on technology upgrades. Empirical studies conducted for the power sector highlight the effect of the EU ETS on encouraging firms to switch from coal to gas to generate electricity (Berghmans et al. 2014; Delarue et al. 2008; Ellerman and McGuinness 2008). However, while this might be considered a case of adoption, it does not represent an actual technology upgrade but rather a reranking of generation technologies, i.e., a merit-order effect.

The diffusion of low-carbon technologies ensures more sustained and lasting emission reductions compared to those achieved through operational changes or output reductions. This is especially true when the technology adopted is zero-carbon rather than just low-carbon (Lilliestam et al. 2022). The EU ETS has been found to reduce emissions in the power sector (Delarue et al. 2008), manufacturing (Abrell et al. 2011; Petrick and Wagner 2014; Wagner et al. 2014; Klemetsen et al. 2020) and aviation (Fageda and Teixidó 2022; Kang et al. 2022). These effects, however, can be reversed if, as in the case of aviation, an external shock causes an increase in the supply of flights. Emission reductions are likely to be more sustained when they involve technology changes that go beyond operational changes or output reductions. When an airline renews its aircraft fleet with more efficient planes, the emission reductions achieved cannot be reversed,[5] especially if the new technology used is carbon-free.

2.2 Induced Technology Diffusion and the Aviation Sector

As of today, some potentially applicable disruptive technologies, including laminar flow control, open-rotor propulsion, and double-bubble designs, have still to overcome major technical and economic obstacles, which means they are not viable options for commercial aviation in the immediate future (Graham et al. 2014). Likewise, the use of green hydrogen is a long way from being technically feasible.

The two main channels for reducing aircraft emission intensity over the next few years hinge on advancements in aircraft technology and the use of biofuels. Nonetheless, biofuels face significant challenges, including high CO_2 abatement costs, steeper price increases compared to jet fuel, a lack of sufficient feedstock, and the need for major capital investment in bio-refining infrastructure (Staples et al. 2018).[6] Altogether, these challenges have resulted in biofuels accounting for less than 0.1% of total aviation fuel consumption. This leaves retrofitting options and in-time aircraft substitution as more rational options for the sector, even though their fleets continue to depend on petroleum-based fuels.

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[5] Unless there is a rebound effect triggered by this higher efficiency which makes flying cheaper, hence, creating an increase in demand (the so-called Jevons paradox). However, this is not the case as the number of flights has fallen compared to the counterfactual (Fageda and Teixidó 2022; Kang et al. 2022).

[6] The production costs of biofuels are estimated at 3 to 6 times the market price of fossil aviation fuels (European Commission 2021). Pavlenko et al. (2019) estimates that the most cost-effective biofuel to reduce CO_2 emissions in aviation uses vegetable oils (with abatement costs at approximately €200 per CO_2/ton). However, competition from other sectors (like road transport) leads to an upward trend in prices and will ultimately lead to supply constraints. The next most cost-effective options are the gasification of municipal solid waste and lignocellulosic feedstocks with abatement costs of approximately €400–500 per CO_2/ton.

As fuel consumption represents between 20% and 30% of airlines' total operating costs, there are strong economic incentives to operate with the most efficient aircraft possible. In practice, the ETS represents an increase in fuel costs, so the airlines most affected by the trading system are likely to have even stronger incentives to operate with more efficient aircraft (Brueckner and Zhang 2010). In this regard, there has been an abrupt increase in CO_2 prices in recent years. In 2013, the mean CO_2 price was €4.45 per ton, increasing from €24.84 per ton in 2019 to €53.55 per ton in 2021 (reaching €88 by the end of that year).[7]

By way of illustration, Fig. 1 shows the cost of CO_2 per aircraft for selected narrowbody aircraft using 2021 prices and 1,500 km.[8] Considering that a single aircraft in this distance range usually makes an average of four flights per day, the total CO_2 costs for airlines are far from negligible. For example, the difference in CO_2 costs between using an A320 and its new, more efficient version, the A320neo, is approximately €270,000 per year for a single aircraft. Regarding the installation of winglets, at a cost of around €800,000, the difference in CO_2 costs between a B737-800 for a single aircraft with and without winglets is around €98,000 per year.[9] Fuel savings are higher, but these figures suggest that the EU ETS provides an additional incentive to use more efficient aircraft and/or install winglets in the current fleet.

As a result, as the theory predicts, airlines affected by the EU ETS may have greater incentives to reduce aircraft emission intensities and, hence, use newer, more efficient

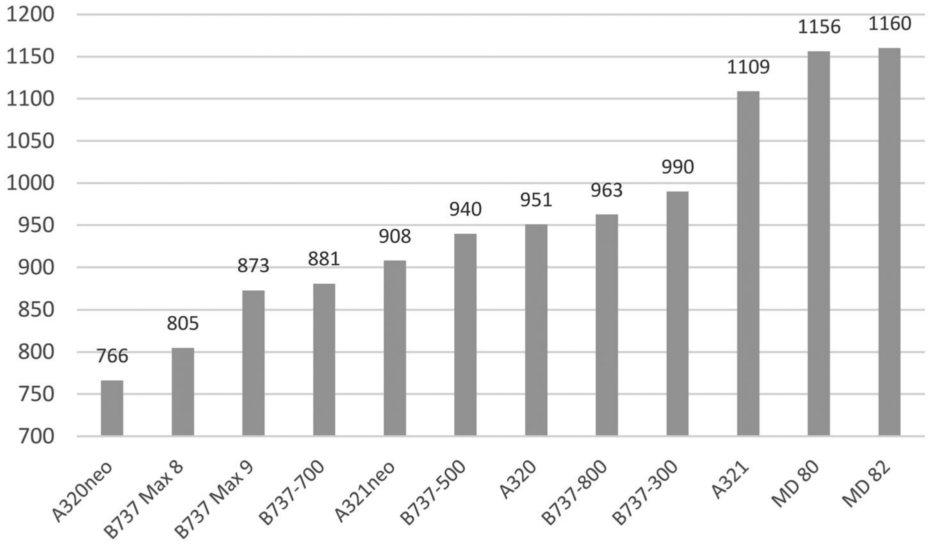

Fig. 1 CO_2 costs per selected narrowbody aircraft. Note: This figure plots CO_2 costs (euros per flight) for a flight of 1,500 km and the average carbon price in 2021. Source: RDC aviation, Eurocontrol

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[7] See Fig. A3 in the appendix for a detailed evolution of carbon prices in the EU ETS.

[8] Mainline jets include narrowbody (with a single aisle of seats) and widebody (two or more aisles of seats) airplanes. Regional planes have a single aisle of seats, but they are smaller than the ones we call narrowbody (with fewer than 100 seats) and include regional jets and turboprops. Narrowbody aircraft account for around 70–80% of flights in the European market.

[9] Note that the fuel-efficiency benefits of winglets vary depending on aircraft age and flight profile. It should be also mentioned that another advantage of winglets is noise reduction.

planes or, at least, the most efficient models for a particular distance range. Similarly, the EU ETS may spur airlines to withdraw old aircraft at an earlier date or, to absorb traffic growth with more efficient aircraft. Finally, the retrofitting of existing aircraft, including the installation of winglets (or sharklets), may be more intensive on ETS routes. Moreover, this has the advantage of not requiring the substitution of aircraft.

In the following sections, we empirically analyze the extent to which carbon pricing has affected the adoption of available technology, in particular, aircraft replacement and fitting winglets.

3 Data and Methods

We use annual data for the period 2010 to 2021 for all flights within Europe at the aircraft-airline-route-year level, where the route is the airport-pair. We record, at this level, the total number of seats, frequencies, aircraft size, distance flown, the operating airline, and the aircraft model used (including whether winglets and sharklets are fitted).[10] Data have been obtained from RDC Aviation (Apex schedules). To estimate emissions at the airline-route level we use Eurocontrol's small emitter tool (SET), designed to assist aircraft operators in their monitoring and reporting obligations for the EU ETS. The SET is based on fuel burn samples of real flight operations and provides accurate estimates of emissions for any given distance and aircraft model. With the aircraft model, the number of seats per aircraft and the route distance, we can estimate emissions per seat-km at the airline-route level per year.[11]

We have also collected urban population data for the points of origin and destination of all routes. For urban areas with more than 300,000 inhabitants, data have been obtained from the UN's World Urbanization Prospects database. For urban areas in the European Economic Area, Switzerland and Turkey with fewer than 300,000 inhabitants, we have collected data from Eurostat (NUTS 3). For urban areas in the remaining countries with fewer than 300,000 inhabitants, we have collected data from their respective national statistics agencies. We also consider income per capita at both endpoints of the routes at the country level. Data has been obtained from the World Bank Development Indicators database. Additionally, we have used supply data to construct an indicator of the intensity of competition, namely the Herfindahl-Hirschman index (HHI), which is based on the sum of the square of the share of flights of airlines operating a route. Moreover, a dummy variable taking a value of one is included for city-pairs with high-speed rail services (speeds over 200 km/hour). We obtained information about each line from the International Union of Railways.

The routes considered may be operated by different types of airlines, primarily low-cost or network carriers. Since all regressions include airline fixed effects, the influence of the airline business model is captured by those fixed effects. Similarly, the impact of joint venture agreements –signed either in 2009 or 2010—is also captured by the airline's fixed effects. In addition, we include a variable accounting for the size of the airline operating the

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[10] Unfortunately, data for cargo flights are not available. However, note that a significant amount of cargo is handled by passenger flights.

[11] When different aircraft are used by an airline on the same route, we compute the corresponding mean values weighted by the number of flights made by each aircraft. Low-cost airlines generally operate routes with a single aircraft model, but it is usual that network airlines operate with different models on the same route.

route in a given year, defined as the total number of flights operated by that airline in that year.

The analysis of the choice of aircraft is based on a sample at the aircraft-airline-route-year level, totaling 632,377 observations. The analysis of aircraft emission intensity is based on a sample at the airline-route-year level (as for emission intensity, we do not need to split airline-route level observation in terms of aircraft). Here, we have collected data for 80,698 airline-route pairs, totaling 333,023 observations. To minimize the distortions of those flights that respond to contingent or circumstantial events, we restrict our sample to airline-route pairs with at least one flight per week. Since some airline-route pairs do not have a flight or have fewer than one flight per week in a particular year, our resulting dataset is an unbalanced panel of 530,162 observations when the sample used is at the aircraft-airline-route level and 158,891 observations when sampled at the airline-route level.

Narrowbody aircraft, the most popular type of aircraft on medium haul routes, dominates the European market. In contrast, widebody aircraft only have a marginal presence, given that they are mainly used for long-haul flights. Smaller regional aircraft, in contrast, are only used for short-haul routes. In this regard, the fuel-efficiency of narrowbody aircraft improves with distance, implying that regional aircraft may be environmentally preferable on very short-haul routes.

Newer aircraft are, in general, more efficient than older planes, tending to have lighter airframes, improved aerodynamics, and more efficient engines (Abrantes et al. 2021; Graham et al. 2014; Bravo et al. 2022). The Airbus A-320 and the Boeing B737-800 are the two most frequently used planes in Europe, accounting together for 75% of aircraft in Europe. Their new generation models –i.e., the A-320neo and the B737-MAX– generate, respectively, 31% and 18% fewer kg of CO_2 per seat-km than their older versions (A320ceo, B737-800).[12] Older models that were popular until the early years of this decade, such as the MD-80, generated 24% more emissions per seat-km than a comparable model such as the A319. Yet, before substituting an aircraft, airlines may consider the available retrofit measures. One such measure is fitting winglets, which have a marked impact on a plane's aerodynamics. As a result of both retrofitting and aircraft substitution, the use of winglets has increased tenfold from 2010 to 2021, with 30% of aircraft using winglets by the end of this period.

Due simply to the gradual replacement of aging aircraft, the overall trend in the European aviation market—across both ETS and non-ETS routes—shows a steady decline in emissions intensity. Figure A1 in the appendix shows a 16% improvement in average emissions intensity from 2010 to 2021. However, it is important to note that total emissions depend on both emissions per seat-km and total seat-km flown (which continue to grow). Despite gains in efficiency, sustained traffic growth has continued to drive up overall emissions, only marginally offset by lower emissions intensity (EEA 2024).

A clear trend, therefore, emerges as flying has become less carbon intensive over the years. Figure 2 shows boxplots comparing emission intensities in 2010 and 2021: the minimum values, i.e. the emission intensity of the cleanest aircraft, are around 0.05 kg of CO_2 per seat-km in both years. The upper values, in contrast, together with the upper quartiles, show a more prominent downward shift, bringing down the median emission intensity val-

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[12] In the case of the B737-800, the use of winglets by default had already implied a great improvement in efficiency.

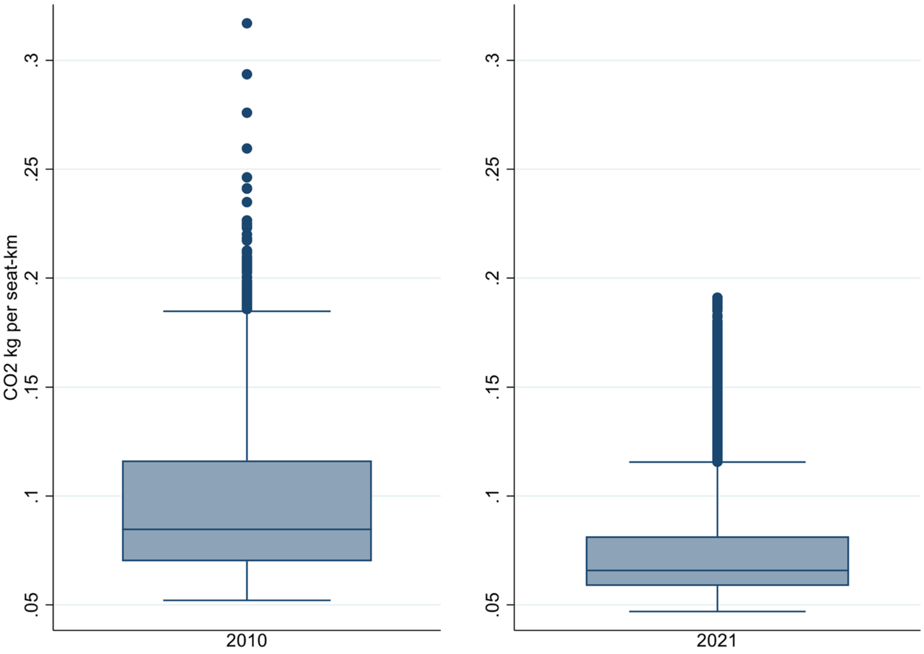

Fig. 2 Emissions per seat-km (2021 vs. 2010). Notes: All outliers are excluded (that is, 1% maximum and minimum values). Data refers to the European market, including treated and control routes

ues. As such, the trend in emission intensity appears to be mainly attributable to airlines moving towards the technology frontier rather than shifting it forward.

Technology change in the sector has never been disruptive, rather it has focused on the emergence of improved versions of existing models. Within this context, the goal of our empirical strategy is to identify the effect of carbon pricing on this overall pattern. In other words, we aim to isolate the effect of the EU ETS from the sector trend described above.

4 Identification Strategy

The way in which the EU ETS was rolled out within the aviation sector is central to our identification strategy, and fundamental for interpreting our results. Under Directive 2008/101/EC of the European Parliament and of the Council of 19 November 2008, the EU decided to include CO_2 emissions from aviation within its ETS. This meant that, as of January 1, 2012, all flights from, to, or within Europe –i.e., landing or taking off from an airport in the European Economic Area– were to be regulated under the carbon trading system and, hence, all airlines, regardless of their nationality, would require allowances for every ton of CO_2 emitted during the year. This generated unprecedented controversy: international carriers refused to adhere to European law and some countries even forbade their airlines from complying with the EU ETS. US carriers challenged the Directive in the EU Courts arguing that the policy infringed national sovereignty and international agreements. Although the Court rejected these claims, more international pressure was brought to bear. Several Latin American countries, Japan, India, Mexico, Russia, and China signed a joint declaration opposing the inclusion of international aviation in the EU ETS. In November 2012, in response to this pressure, the EU Commission decided to reduce the scope of the EU ETS to flights within the EEA, applying this change retrospectively to 2012 flights. The decision was formally adopted in April 2013, before airlines were required to surrender allowances for 2012. As a result, free allowance allocations exceeded verified emissions from the aviation sector in 2012, and airlines effectively did not need to buy additional allowances (European Environment Agency [EEA], 2024). [13]

This policy change allows us to compare ETS flights to comparable non-ETS flights. Since legislative uncertainty was resolved and the regulation was fully implemented only from 2013 onward, we treat 2013 as the first effective year of EU ETS implementation for the aviation sector. As described above, the EU ETS was initially designed to cover all routes departing or landing in an EEA airport. In this regard, it is very unlikely that aircraft choices and associated emissions intensity in specific route-airline pairs would have followed different trajectories between EEA and non-EEA flights absent the EU ETS (parallel trends assumption), especially given the technological context of the aviation sector, where almost three-quarters of aircraft used in Europe are manufactured by just two companies, Airbus and Boeing. As a result, airlines across countries exhibit highly homogeneous aircraft technology, making the parallel trends assumption particularly reasonable in our context.

We apply the logic of difference-in-differences, a common methodology adopted within the treatment evaluation framework (Angrist and Pischke 2009; Gertler et al. 2016). Thus, routes affected by the EU ETS are all those connecting two airports within the EEA since 2013, while our control routes include those that connect an EEA airport with a non-EEA airport and those that connect two non-EEA airports. The EEA countries include the EU28, Norway and Iceland. European non-EEA countries include Albania, Armenia, Azerbaijan, Belarus, Bosnia and Herzegovina, Georgia, Macedonia (FAROM), Montenegro, Moldova, Russia, Serbia, Switzerland, Turkey, and Ukraine (see Fig. 3). As we explain below in more detail, we also run additional analyses in which we split the sample between Eastern and Western Europe.

We estimate the following equations for aircraft *a* and route-airline pair *i* in year *t*:

$$\log(emissions\_intensity)_{it} = \alpha\ + \beta\, ETS_{it} + \lambda\, X_{it} + \gamma_i + \eta_t + \epsilon_{it} \tag{1}$$

$$aircraft\_type_{ait} = \alpha\ + \beta\, ETS_{it} + \lambda\, X_{it} + \gamma_i + \eta_t + \epsilon_{ait} \tag{2}$$

where *emissions_intensity* refers to CO_2 kgs per seat-km, and *aircraft_type* to dummies for narrowbody, fuel-efficient aircraft or winglet installation.[14]

ETS is a dummy identifying treated flights since 2013. It is built by interacting a dummy variable for within-EEA flights and a dummy for 2013 and beyond. *X* is a set of control covariates: population and income of the points of origin and destination of the route, the

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[13] While opportunity costs have been shown to influence firm behavior in several sectors (e.g., Fabra and Reguant 2014), it is important to note that although airlines could purchase allowances from any sector, the aviation allowances allocated for free in 2012 could only be used within the aviation sector. This greatly limited the relevance of opportunity costs in that year.

[14] In Eq. (1), the unit of observation is the route-airline pair (e.g., Lufthansa flights from Frankfurt to Barcelona. In Eq. (2), the unit of observation is the aircraft-route-airline pair (e.g., Lufthansa flights from Frankfurt to Barcelona operated with an A320). Winglets also include sharklets.

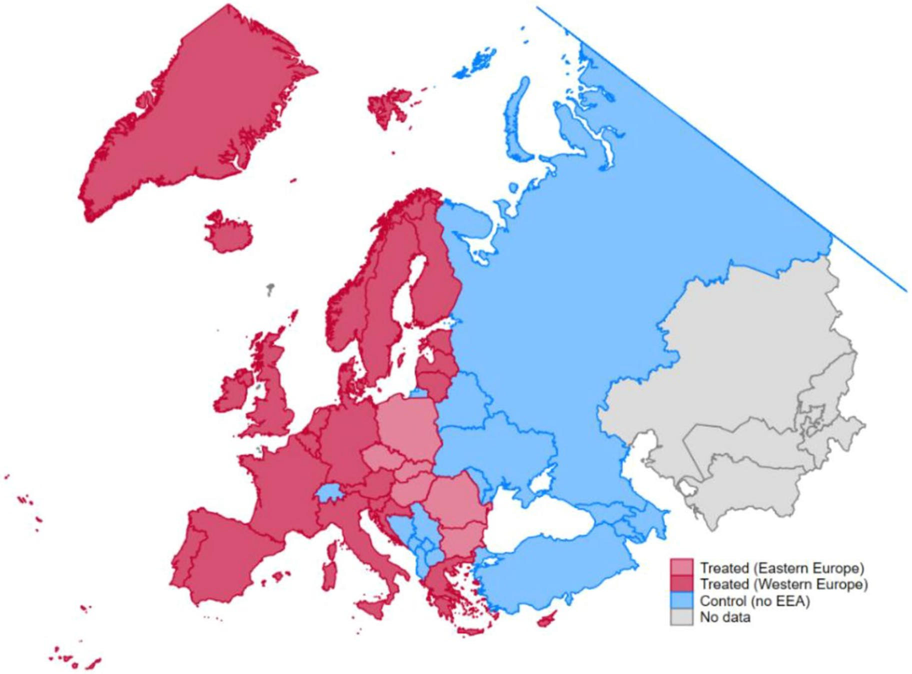

Fig. 3 Map with treated and control countries. Notes: This map shows countries in the non-European Economic Area-EEA (in blue) and treated countries in the Eastern sample (light red) and Western sample (dark red). Treated routes are those flying *within* the EEA countries

HHI index, a dummy for high-speed rail services, and airlines size. For our emission intensity regressions, we also control for the average size of the aircraft.

The control variables are motivated as follows: More populous or affluent areas may justify the use of different aircraft types if these characteristics drive higher demand for air travel. Furthermore, airlines in richer countries may have more resources to invest in newer aircraft. The Herfindahl-Hirschman Index (HHI) captures market concentration as a proxy for competition intensity; greater competition could incentivize airlines to use more fuel-efficient aircraft to reduce costs per seat-kilometer. Similarly, routes competing with high-speed rail may lead airlines to strategically select different aircraft types, particularly given that these are typically shorter routes. Finally, airline size accounts for the fact that larger airlines tend to operate a more diverse and potentially older fleet.

Emissions intensity in our data reflects several technological and operational choices made by airlines, particularly regarding aircraft type, seat configuration, and flight distance. Larger aircraft, denser seating, and longer routes are generally associated with lower fuel use per seat-kilometer (Csereklyei and Stern 2020; Lo et al. 2020; Park and O'Kelly 2014). These factors are observable in our dataset. However, other strategies that can reduce emissions intensity—such as slower cruising speeds, improved load factors, or optimized flight altitudes—are not captured in our data (Brueckner et al. 2024; Fageda and Oesignmann, 2025; Ekici et al. 2024). This should be kept in mind when interpreting the estimates.

All continuous variables are in logs. We add airline-route fixed effects and year fixed effects.[15] Note that route-airline fixed effects capture the distance effect, the type of airline (either network or low-cost), and other time-invariant factors related to the characteristics of the airport or the airline. For instance, the impact of joint venture agreements –signed either in 2009 or 2010—is also captured by the airline's fixed effects. Standard errors are robust to heteroscedasticity and clustered at the route level. We apply weights based on the number of flights for each aircraft-airline-route or route-airline combination. This allows us to give more weight to those route-airline pairs with more flights, which are thus those route-airline pairs that generate more emissions in absolute values.

Table 1 (and figure A2 in the appendix) show the mean differences between treated and control routes for all covariates (and emissions intensity) used in the empirical analysis in the pre-treatment period (2010–2012). While we do not find statistical differences in terms of emissions intensity, treated routes link richer but less populated endpoints. Also, airlines in treated routes are bigger. Differences with other covariates are more modest. Key to difference-in-differences identification strategy is that the parallel trend assumption holds. These covariate differences between treated and control flights, while informative, are only relevant in terms of identification strategy if parallel trends assumption depend on them, i.e., conditional parallel trends.[16] As a robustness check, we also implement the synthetic difference-in-differences method (Arkhangelsky et al. 2021), which optimally weights the control group to balance covariates and ensure parallel pre-treatment trends between treated and control groups.

We evaluate the plausibility of parallel trends assumption by implementing an augmented difference-in-differences estimator. This involves estimating the impact of the treatment in different years of the sample period (Autor 2003), by applying the following equations:

Table 1 Mean differences between treated and control (pre-treatment period)

	Description	Treated	Control	T-test
Emissions_intensity	CO_2 kgs per seat-km	-2.429	-2.425	-0.98
Income	Average income at the country level of origin and destination of the route	10.543	10.07	78.84***
Airlines_size	Total flights of the airline in the airline-route pair	11.411	10.444	46.75***
HSR	Dummy variable for routes with non-stop high speed rail services	0.011	0	8.61***
HHI	Hirshman-Herfindahl index at the route level in terms of flights	0.685	0.671	3.8***
Aircraft_seats	Average seats of aircraft employed in the airline-route pair	4.881	4.864	2.26**
Population	Average population at the urban level of origin and destinations of the route	13.918	14.384	-34.68***

Note: This table shows the average differences between treated and control routes before 2013. All variables are in logarithms except HHI and HSR

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[15] In the regressions where the dependent variable is the aircraft type, we do not consider aircraft-type fixed effects because the model is over-identified with an R^2 of 1.

[16] Using time-varying covariates when estimating a difference-in-differences model estimated via two-way fixed effects, our case here, implies assuming covariates are not affected by the treatment (i.e. not bad controls) and that there are no covariate-specific trends in both groups (Sant'Anna and Zhao 2020). We report regressions without control covariates in Table A1 of the Appendix and find that the estimates are not significantly different.

$$\log(emissions\_intensity)_{it} = \alpha + \beta_k(EEA_i \times t) + \lambda X_{it} + \gamma_i + \eta_t + \epsilon_{it} \tag{3}$$

$$aircraft\_type_{ait} = \alpha + \beta_k(EEA_i \times t) + \lambda X_{it} + \gamma_i + \eta_t + \epsilon_{ait} \tag{4}$$

where *EEA* is a dummy variable equal to one when the route is in the treated group, and *t* is the year. Thus, β is the coefficient on the treatment effect in year *t*. The non-significance of the coefficients in the years before 2013 adds plausibility to the common trend assumption in the pre-treatment period, while their significance after 2013 is indicative of the durability of the effect over time. Note here that airline decisions on aircraft are not made on a year-by-year basis and the average order-delivery time for narrow and wide body jet aircraft is around 2 years (Dray 2013).

Figure 4 shows that the treated and control groups were trending similarly for the key variables in our analysis before the policy implementation in 2013, that is, emissions per seat-km (top-left panel), share of the most fuel-efficient aircraft model, both at the full sample and per year and distance levels (top-right and bottom-left panel), and share of aircraft with winglets fixed (bottom-left panel). This further adds plausibility of parallel trends.

This latter descriptive evidence already suggests that the effect of the EU ETS is especially notable in terms of winglet installation. Note also that Fig. 4 indicates a worsening of emissions intensity in the years of the pandemic, potentially because of the higher use of small planes that are less efficient than larger ones. Interestingly, this trend seems to affect

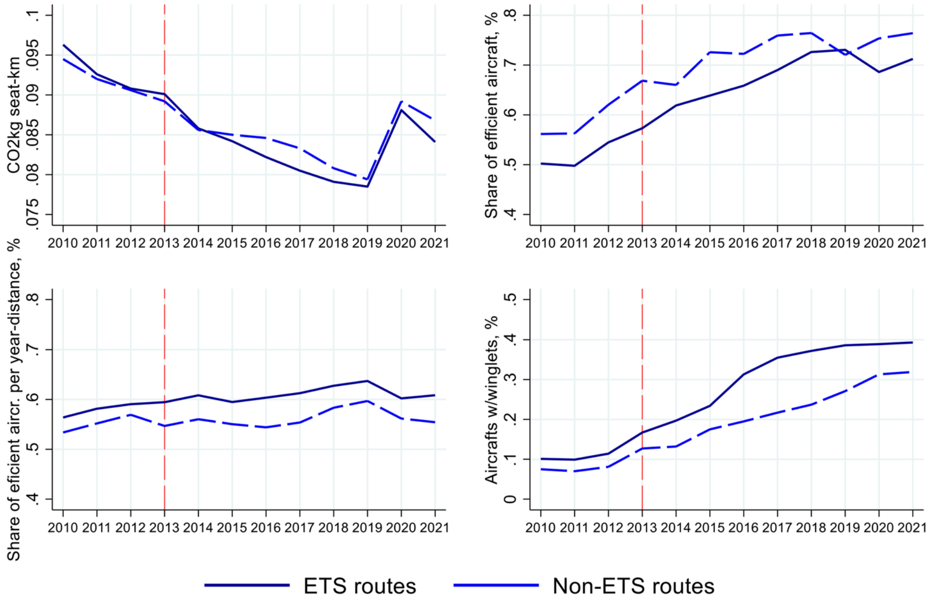

Fig. 4 Evolution of emission intensities and shares of aircraft used. Notes: Mean values for the different dependent variables used in the difference-in-differences estimator (solid lines for treated and dashed for control routes). Efficient aircraft denote aircraft with emissions per seat-km below the sample average, while efficient aircraft per year-distance are aircraft with emission intensity per seat-km below the average in a given year and distance. The relatively constant gap between treated and control airline-routes before the treatment adds credibility to the parallel trends assumption, i.e. had the policy not been in place, the trends would have continued as before 2013

treated and control routes equally, further suggesting that treated and control groups share the underlying market and technological conditions.

Moreover, to generate valid causal estimates, difference-in-differences require the stable unit treatment assumption (SUTVA) to be met. If airlines moved their older and less efficient aircraft from ETS regulated to non-regulated routes, then our estimates would be biased as the control group would increase their emission intensities because of the EU ETS. However, as Fig. 4 clearly shows, both treated and control routes reduced their emission intensities while increasing their share of efficient planes across the period. In addition, retrofitting activities (i.e., winglets) also have similar improving trends. This shows that a process of improvement of the aircraft fleet is taking place both inside and outside the emissions market.

5 Results

5.1 The Effect of EU ETS on the Emission Intensity of Aircraft

Table 2 shows the results of the regressions in which the emission intensity of the aircraft operating the route is the dependent variable[17]. In column 1, we show the results using the unweighted sample. In column 2, we show the results of the baseline regression. To account for the potential distortion of the COVID-19 pandemic, column 3 focuses on the period 2010–2019. In column 4, we exclude the year 2012 due to the uncertainty about which

Table 2 EU ETS and emission intensity (emissions per seat-km)

Variables	(1)	(2)	(3)	(4)	(5)
ETS	-0.0185***	-0.0212***	-0.0203***	-0.0229***	-0.0418***
	(0.00291)	(0.00318)	(0.00314)	(0.00355)	(0.00505)
Observations	158,891	158,891	131,807	146,765	136,238
R-squared	0.975	0.973	0.977	0.973	0.974
Route-airl. FE	YES	YES	YES	YES	YES
Covariates	YES	YES	YES	YES	YES
Year FE	YES	YES	YES	YES	YES
Sample	All	All	All	All	Sutva
Period	2010–2021	2010–2021	2010–2019	2010–2021	2010–2021
Year 2012	YES	YES	YES	NO	YES
Clusters	Route	Route	Route	Route	Route
Weights	NONE	Flights	Flights	Flights	Flights

Note: This table shows difference-in-differences coefficients using as our outcome variable a measure of flight emission intensities (emissions per seat-km). Column (1) shows the results without applying weights, while the rest of columns show the results with the sample weighted by the number of flights. To check if the COVID-19 pandemic may distort our results, column (3) considers the period 2010–2019. To see if the effect could be affected by the uncertainty in 2012, column (4) shows results when excluding this year. In column (5), the control group only includes those routes flying within non-EEA countries to check whether the potential violation of SUTVA assumption could distort our results. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

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[17] Table A1 in the appendix shows these same regressions showing covariate coefficients. Furthermore, it shows a regression without controls to examine how their inclusion influences the results.

routes were affected by the EU ETS in that year. We find that emissions per seat-km on treated routes are about 2% lower than in the counterfactual.

In column 5, we show the results for alternative control group as a robustness check. It could be that the improvement found in emission intensity is the result of airlines moving their older (or less efficient) aircraft to routes that connect an EEA airport with a non-EEA airport, i.e. to flights in the control group. This would violate the SUTVA assumption and yield biased estimates. More specifically, previous results would be overestimated, as they would capture the reduction of treated routes (obtaining better planes) plus the increase in control routes (being assigned the older planes from the treated routes). Hence, column 5 shows estimates using a sample in which the control group excludes routes that link an EEA with a non-EEA airport, i.e. the control group is limited to non-EEA airports only. The logic of this is that EEA airlines may simultaneously operate within-EEA routes and routes that link an EEA with a non-EEA airport. However, it is, in fact, very unusual that these EEA airlines operate routes connecting two non-EEA countries: specifically, less than 1% of their flights. As a result, restricting the control group to non-EEA airports greatly reduces these potential SUTVA violations and, therefore, if our results were overestimated, we should see a smaller effect in column 5. For example, Lufthansa could move aircraft from the Frankfurt-Istanbul (control) route to the Frankfurt-Barcelona (treated) route because it operates both routes. However, Lufthansa (and the other EEA airlines) rarely operate on routes connecting non-EEA airports (e.g. Istanbul-Moscow). Thus, the issue related to the possible deployment of aircraft from treated to control routes is not applicable to the sample case which restricts control routes to within non-EEA routes. If, on the other hand, airlines in this non-EEA group purchased airplanes in a secondhand market, then they would not buy less efficient aircraft than the ones to substitute, hence, this would still be improving the non-EEA aircraft in which case we would still be underestimating the effect.

We find, however, a larger EU ETS effect, with about 4% lower emissions per seat-km, indicating that no strategic movement of planes takes place within or between airlines. This greater effect, however, also reveals that a higher degree of heterogeneity between the treated and (restricted) control groups affect size effects, suggesting potential heterogenous effects that we further explore below.

Figure 5 shows the event-study estimates of these prior effects illustrating both dynamic effects and the plausibility of the common trends assumption. Point estimates have risen to about 3%-6% since 2015. This jump in emission intensity can be considered a telltale sign if we consider that aircraft investment is not immediate and that a few years must elapse between a purchase decision and aircraft use. At this point, note that the pre-treatment period (conditioned by data availability) is relatively short but, in practice, treated and control routes follow similar trends until 2014. The event-study estimates also show that the uncertainty in 2012 had no effect on emissions intensity and therefore on airline aircraft choices.

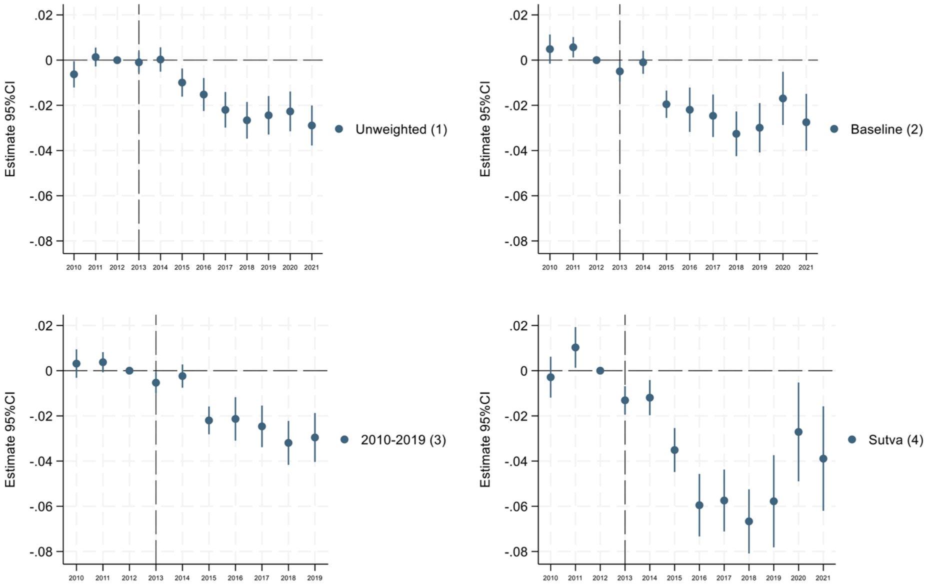

Fig. 5 Event-study estimates of the EU ETS effect on flight emission intensity. Notes: This figure plots the results from an event-study analysis of the differences in emission intensity between the EU ETS regulated air-routes and other comparable air-routes before and after policy implementation. The coefficients reported are derived from Eq. (3) in which we interact the treatment variable with year indicators. The top-left panel shows the results for the unweighted sample, while the top-right panel shows the results of the baseline regression applying weights in terms of total flights. The bottom-left panel considers the period 2010–2019, while the bottom-right panel uses the sample with the control group restricted to non-within-EEA routes. 2012 is used as a reference year. The confidence interval is set at 95% and standard errors are clustered at the route level)

5.2 Heterogenous Effects: Western and Eastern Europe

Our first results table (Table 2) show that, when we limit the control group to flights between non-EEA airports, the ETS effect increases from −2% to -4%. This higher effect, besides ruling out potential SUTVA violation (which would have resulted in a lower effect, not a higher one), also reveals potential heterogeneous effects stemming from the fact that most non-EEA airports are in Eastern Europe. To address these potential heterogenous effects, we consider additional regressions based on the historical and economic East-West divide in Europe (see Fig. 3).

For Western Europe regressions, the treated group consists of flights within the 15 pre-2004 EU Member States,[18] Iceland, and Norway, using flights from/to Switzerland as the control group. Switzerland is not an EEA country but is associated with the European Union through a series of bilateral agreements and is part of the single market. As such, Swiss flights are non-treated flights up to 2020, when the Swiss ETS was linked to the EU ETS. We therefore need to limit the sampled period here from 2010 to 2019.

For Eastern Europe regressions, we exclude flights from/to the 15 pre-2004 EU Member States, Iceland and Norway from the treated group, and flights from/to Switzerland from the

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[18] Austria, Belgium, Denmark, Finland, France, Germany, Greece, Ireland, Italy, Luxembourg, the Netherlands, Portugal, Spain, Sweden and the United Kingdom.

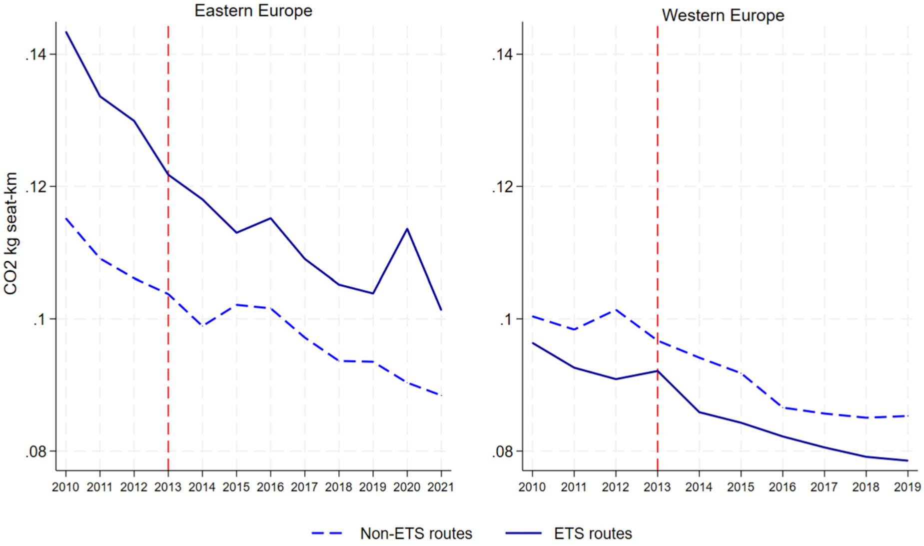

Fig. 6 Evolution of emission intensities (Eastern vs. Western sample). Notes: Mean values for emissions intensity in treated and control routes. The left panel is for the Eastern sample, while the right panel is for the Western sample. Treated routes have higher emissions intensity than control routes in the Eastern sample, while the opposite is for the Western sample

control group. This means the treated group consists of routes in Bulgaria, Croatia, Cyprus, Czech Republic, Estonia, Hungary, Latvia, Lithuania, Malta, Poland, Romania, Slovakia and Slovenia. The control group includes Albania, Armenia, Azerbaijan, Belarus, Bosnia and Herzegovina, Georgia, Macedonia (FAROM), Montenegro, Moldova, Russia, Serbia, Turkey, and Ukraine.

Figure 6 shows the evolution in emissions intensity in both treated and control routes for the Eastern and Western sample, with a decreasing trend in all cases. Interestingly, emissions intensity in the control group has similar levels in both samples. In contrast, emissions intensity is higher in the treated group in the Eastern sample, and it is lower in the treated group in the Western sample. This allows us to explore the airline responses to the ETS in a different setting, at least in terms of the initial emissions intensity.

Table 3 and Fig. 7 show the results of the additional regressions based on the historical and economic East-West divide in Europe. Note that we also report results for the restricted control group sample that only include within non-EEA routes.

The ETS effect for Eastern Europe was substantially higher than the average treatment effect in the full sample. The average effect is about −10% and reaching point estimates of around −15% three year after implementation (Fig. 7). The average effect is non-significant for Western economies with a rather ambiguous trend before and after implementation of ETS. One potential explanation for the differential effect is that airlines operating Eastern European regulated routes were further from the technological frontier, providing more opportunity to improve emission intensities. Note also that a potential deployment of aircraft from treated to control routes is unlikely in the Eastern sample, even without restricting the control group, given that emissions intensity is lower in the control group.

Table 3 EU ETS and emission intensity (emissions per seat-km). Eastern vs. Western Europe

Variables	(1)	(2)	(3)	(4)
ETS	-0.0956***	-0.0994***	0.00381	-0.0117
	(0.0179)	(0.0185)	(0.00397)	(0.0113)
Observations	15,098	12,131	113,917	108,365
R-squared	0.957	0.960	0.980	0.981
Route-airline FE	YES	YES	YES	YES
Year FE	YES	YES	YES	YES
Sample	Eastern	Eastern-SUTVA	Western	Western-SUTVA
Period	2010–2021	2010–2021	2010–2019	2010–2019
Clusters	Route	Route	Route	Route
Controls	All	All	All	All

Note: This table shows difference-in-differences coefficients using as our outcome variable a measure of flight emission intensities (emissions per seat-km). In column (1), the restricted sample excludes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland. In column (2), the same sample is considered but the control group only includes those routes flying within non-EEA countries. In column (3), the restricted sample only includes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland, while column (4) as additional restriction that control routes are within non-EEA countries. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

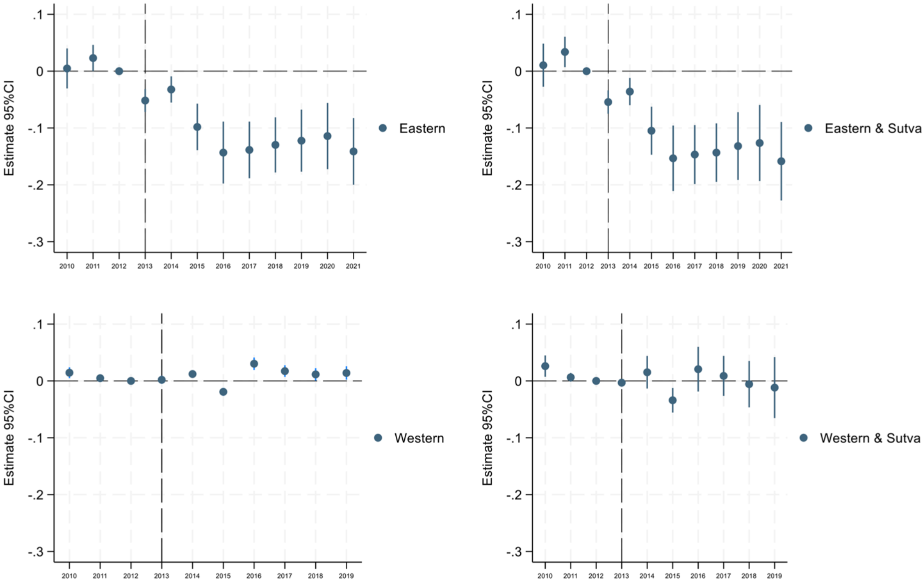

Fig. 7 Event-study estimates of the EU ETS effect on flight emission intensity (Eastern vs. Western Europe). Notes: This figure plots the results from an event-study analysis of the differences in emission intensity between the EU ETS regulated air-routes and other comparable air-routes before and after policy implementation. The coefficients reported are derived from Eq. (3) in which we interact the treatment variable with year indicators. The top panels show the results for the sample that excludes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Eastern Europe). The bottom panels show the results when the sample only includes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Western Europe). 2012 is used as a reference year. The confidence interval is set at 95% and standard errors are clustered at the route level

This differential effect between Eastern and Western European countries is consistent with the larger effect we find for the sample with the restricted control group to routes flying within non-EEA countries (SUTVA sample). The weight of Switzerland in the control group in this subsample is much smaller given that most of flights from/to Switzerland involve EEA countries. As a result, the estimates from the SUTVA sample capture part of the greater EU ETS effect in the Eastern countries' routes.

5.3 Synthetic Difference-in-Differences

As a robustness check, we also run regressions using the synthetic difference-in-differences method, an estimator developed by Arkhangelsky et al. (2021) that combines features of both difference-in-differences and synthetic control methods (Abadie 2021). Synthetic difference-in-differences still calculate the treatment effect as the difference before and after treatment between treated units and control units. However, the control units are optimally re-weighted and matched in terms of observed and unobserved time-varying covariates (synthetic control units) to ensure parallel trends before the policy implementation and, in general, a better fit for the control group. In this regard, the synthetic difference-in-differences method correct for differences across covariates between treated and control routes.

As our heterogeneity analysis already suggests, the treatment and control groups consist of very different countries, with treated countries having higher income compared to most control countries (except for Switzerland). By splitting the sample into Eastern and Western routes, we can account for part of these income differences. However, by using synthetic difference-in-differences, the control group is optimally weighted considering income jointly with other observed and non-observed time-varying factors.

Table 4 shows the results using this method, considering the full sample (column 1), the sample that limits the control group to within non-EEA countries (column 2), and

Table 4 EU ETS and emission intensity (emissions per seat-km). Estimates from synthetic difference-in-differences

Variables	(1)	(2)	(3)	(4)
ETS	-0.00910***	-0.03654***	-0.06618***	-0.01028***
	(0.00340)	(0.00538)	(0.0133)	(0.003911)
Observations	73,243	62,580	4,440	42,700
Route airline FE	YES	YES	YES	YES
Year FE	YES	YES	YES	YES
Sample	All	SUTVA	Eastern Europe	Western Europe
Period	2010–2021	2010–2021	2010–2021	2010–2019
Controls	All	All	All	All

Note: This table shows the synthetic difference-in-differences coefficients using as our outcome variable a measure of flight emission intensities (emissions per seat-km). In column (1), results using the full sample are shown. In column (2), the control group only includes those routes flying within non-EEA countries. In column (3), the restricted sample excludes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland. In column (4), the restricted sample only includes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland. Standard errors with bootstrap (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

Eastern and Western routes (columns 3 and 4). Interestingly, results for the full sample (column 1) and results for Western routes (column 4) are similar, with an estimated impact of about − 1%. This may be explained by the fact that, when considering the full sample, this method assigns a greater weight to Switzerland to bring the control units closer to most of the treated units. In this regard, the effect for Western Europe, despite becoming statistically significant after accounting for differences in the covariates between Switzerland and treated countries, remains relatively low compared to results for Eastern Europe. The EU ETS in Eastern Europe reduced emission intensity by -6% on average, which is similar to the effect we find with the standard difference-in-differences, albeit smaller in size.

5.4 Carbon Price Effects on Emission Intensities

Since we have information on the mean carbon prices per year, we can estimate the impact of changes in carbon prices on emissions intensity addressing the intensive effect of the ETS. Table 5 shows the results of regressions that consider carbon prices as the main variable of interest (these are zero for all airlines up to 2012). We consider the baseline model – with variations related to the weighting scheme and the time frame, and regressions that use the restricted control group, and Eastern and Western samples. The ETS price variable is negative and statistically significant in all regressions, except in that using the Western sample. The coefficients in Table 5 capture the marginal effect based on a 1€ increase. While such marginal effect is small, carbon prices climbed from 0€ in the period 2010–2012, to €4 in 2013 and €88 at the end of 2021 (see figure A3 in the appendix). Thus, the price effect is similar to the effect reported in previous regressions.

Table 5 EU ETS prices and emission intensity (emissions per seat-km)

Variables	(1)	(2)	(3)	(4)	(5)	(6)
ETS_price	-0.000449***	-0.000504***	-0.00123***	-0.000423**	-0.00131***	0.000177
	(6.75e-05)	(0.000117)	(0.000200)	(0.000193)	(0.000498)	(0.000227)
Observations	158,891	158,891	131,807	136,238	15,098	113,917
R-squared	0.975	0.972	0.977	0.974	0.956	0.980
Route-airline FE	YES	YES	YES	YES	YES	YES
Year FE	YES	YES	YES	YES	YES	YES
Sample	All	All	All	Sutva	Eastern	Western
Period	2010–2021	2010–2021	2010–2019	2010–2021	2010–2021	2010–2019
Clusters	Route	Route	Route	Route	Route	Route
Weights	NONE	Flights	Flights	Flights	Flights	Flights

Note: This table shows the estimates of a regression where the dependent variable is emissions intensity, and the main covariate is the mean carbon price per year. Columns (1) to (3) report the results for the entire sample with differences in terms of weighting scheme and time frame. Columns (4), (5) and (6) consider the sample that only includes as control routes those that are within non-EEA countries, Eastern and Western samples, respectively. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

5.5 The Effect of the EU ETS on Aircraft Choice and Retrofits

Next, we replicate the analysis, focusing now on how the EU ETS affects the type of aircraft chosen to operate. We are interested in determining whether this improvement in emission intensity in the EU ETS flights has been driven by induced technology changes.

Table 6 shows the estimates for four different dependent variables that identify the characteristics of the aircraft chosen: in column 1, we focus on whether the EU ETS has incentivized a greater use of narrowbody planes.[19] This would mean using larger planes and, hence, retiring smaller CO_2-intensive regional planes. In column 2, we examine the most fuel-efficient aircraft in the full sample (i.e., an aircraft is considered efficient when its emission intensity per seat-km is below the sample average) excluding regional aircraft that are always inefficient. In column 3, we refine the measurement of efficiency by considering the most fuel-efficient aircraft in a given year and over a given distance (i.e., an aircraft is considered efficient when its emission intensity per seat-km is below the average in a given year and distance). The result is the effect of the EU ETS on using the most efficient aircraft available. Finally, column 4 shows the effect on winglet use in the aircraft operated, considering the most popular family of planes (the A320 and B737 –classic and next generation–). As opposed to the former specifications, this is a retrofit action which, as such, does not require the substitution of aircraft. We focus on the most popular aircraft family models to separate the effect of retrofitting (upgrades to the current fleet) from the effect of fleet renewal, as new aircraft models (i.e., A320neo, B737 MAX) usually already have winglets fitted.

Table 6 EU ETS on aircraft choice

Variables	(1) Narrowbody Aircraft	(2) Fuel-efficient aircraft	(3) Per year-distance fuel-efficient aircraft	(4) Winglets in aircraft
ETS	0.0328***	0.0356***	0.0527***	0.104***
	(0.00546)	(0.00891)	(0.0126)	(0.00815)
Observations	530,162	475,580	530,162	409,513
R-squared	0.803	0.529	0.691	0.613
Route-airline FE	YES	YES	YES	YES
Year FE	YES	YES	YES	YES
Sample	All	Large aircrafts	All	Popular aircrafts
Clusters	Route	Route	Route	Route
Controls	All	All	All	All

Note: This table shows difference-in-differences coefficients using as our outcome variable dummies for the type of aircraft used. In column (1), we consider a dummy variable for narrowbody aircraft. In column (2), a dummy for fuel-efficient aircraft, in column (3), for the most fuel-efficient aircraft in a given year and over a given distance, and in column (4), for winglet fittings. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

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[19] As opposed to previous regressions where emissions intensity was the dependent variable, these specifications do not control for aircraft size. Differences in terms of aircraft size need to be controlled for when examining how the EU ETS affected regulated airline routes' carbon intensity. However, when examining the aircraft choice, aircraft size is endogenous and may confound the coefficient of interest (yet coefficients do not change significantly).

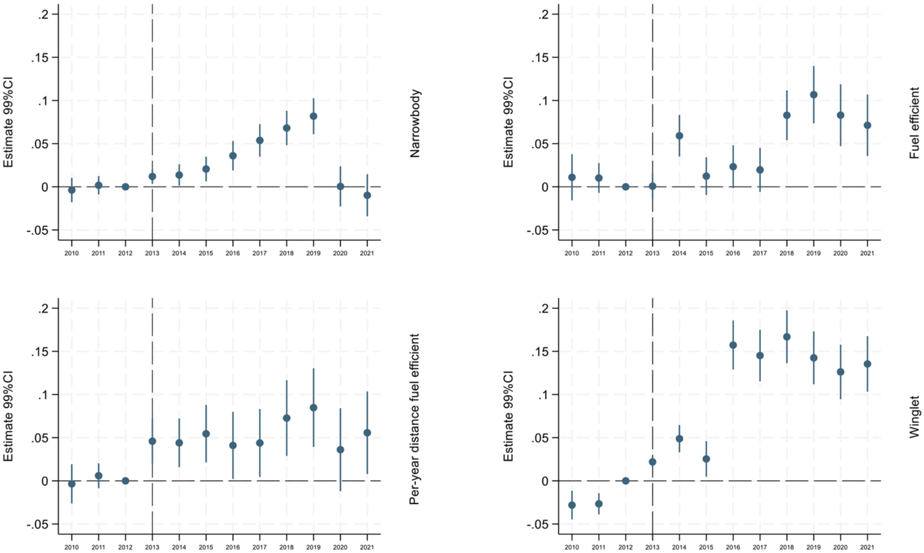

Fig. 8 Event-study estimates for aircraft choice. Notes: This figure plots the results from an event-study analysis of the differences in the type of aircraft used. The coefficients reported are derived from Eq. (4), in which we interact the treatment variable with year indicators. The key assumption is that, prior to policy implementation (red line at 2013), airlines in the control group show no difference from airlines in the treated group in terms of the particular type of aircraft they were operating. This is supported by the fact that the coefficient before policy implementation is not statistically different from zero. The top-left panel shows the annual effects of the EU ETS on the use of narrowbody aircraft, whose emission intensity mainly derives from increasing the size of the plane. The top-right panel shows the effect on the use of fuel-efficient aircraft, defined as those with below-average fuel efficiency. The bottom-left panel shows the effects on the use of fuel-efficient aircraft in a given year and over a given distance. The bottom-right panel shows the annual effects on the use of aircraft with winglets, consisting mainly in a retrofit of the aircraft in use. 2012 is used as a reference year. The confidence interval is set at 95% and standard errors are clustered at the route level

According to our results, the EU ETS has incentivized the use of more efficient aircraft in all categories. Airlines on ETS-routes have a 3% higher probability of using narrowbody aircraft than airlines on non-ETS routes. Similarly, the EU ETS has also increased the use of fuel-efficient planes by 3% and, if we restrict the analysis to efficient aircraft to a given year and distance, the effect is 5%.[20] More striking, however, is the effect on winglet fitting, for which the EU ETS has an average effect of about 10%. Arguably, this higher size effect derives from the fact that this action consists primarily of a retrofit investment and does not require the purchase of a new aircraft.

Figure 8 shows the evolution of the year-on-year estimates of these effects. The EU ETS coefficients prior to the treatment are not statistically significant, which makes the common trends assumption plausible. The figure also shows that the effect on aircraft technology adoption increases over time, with point estimates close to 10% in 2019 in the four aircraft categories. The shock caused by the pandemic slightly slowed this growing trend towards the use of more efficient aircraft on treated routes compared to control routes: there is only a strong distortion when we consider the use of narrowbody aircraft. The pandemic encour-

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[20] Results are essentially the same if we consider that fuel-efficient aircraft are those with less emissions per seat-km than the median sample for the entire period.

aged the use of smaller aircraft on both treated and control routes. In this regard, the results of the regressions in Table 2 suggest that this distortion has not altered the estimated average effect of the EU ETS on emissions per seat-km.

Interestingly, the 2016 jump for winglet fittings (which do not require as much time as the purchase of an aircraft) might be driven by reforms to the EU ETS approved in 2015 –including the market stability reserve (MSR)– taking effect in 2019. This reform involves the introduction of a mechanism to avoid an excess of allowances in the carbon market, considered to be the cause of excessively low prices in the years following the financial crisis. It was expected to affect both levels and the volatility of carbon prices (Borghesi et al. 2023; Bruninx et al. 2020; Perino and Willner 2016), which appears to have been confirmed by recent high and increasing carbon prices. This could eventually further induce the adoption of more efficient aircraft, especially those already to hand, such as winglets.

Table 7 explores the heterogeneity of these technological changes in terms of Eastern and Western Europe routes. Results show that Eastern Europe experienced a remarkable increase in the use of efficient aircraft as a response to the EU ETS: 14% average increase with point estimates well above 20% in 2021 (Fig. 9 top-middle panel), as opposed to Western Europe, where no aircraft upgrade can be identified. In contrast, Western Europe registers a significant change in winglet fitting: 14% on average and reaching point estimates above 20% after 2015 (Fig. 9 bottom-right panel), while, for Eastern Europe, winglet fitting is non-different from zero on average, although an imprecise increase during last sampled years is identified (Fig. 9 top-right panel). The ETS effect on narrowbody planes is not statistically significant

Table 7 EU ETS on aircraft choice (Eastern vs. Western Europe)

Variables	(1)	(2)	(3)	(4)	(5)	(6)
	Narrowbody aircraft	Per year-distance fuel-efficient aircraft	Winglets in aircraft	Narrow-body aircraft	Per year-distance fuel-efficient aircraft	Winglets in aircraft
ETS	0.0139	0.140***	-0.00336	0.00787	-0.00112	0.140***
	(0.0198)	(0.0290)	(0.0229)	(0.00751)	(0.0184)	(0.0146)
Observations	61,553	61,553	43,053	270,353	270,353	130,132
R-squared	0.755	0.724	0.520	0.860	0.717	0.655
Route-airline FE	YES	YES	YES	YES	YES	YES
Year FE	YES	YES	YES	YES	YES	YES
Sample	Eastern	Eastern	Popular aircraft & Eastern	Western	Western	Popular aircraft & Western
Period	2010–2021	2010–2021	2010–2021	2010–2019	2010–2019	2010–2019
Clusters	Route	Route	Route	Route	Route	Route
Controls	All	All	All	All	All	All

Note: This table shows difference-in-differences coefficients using as our outcome variable dummies for the type of aircraft used. In columns (1) and (4), we consider a dummy variable for narrowbody aircraft, in columns (2) and (5), for the most fuel-efficient aircraft in a given year and over a given distance, and in column (3) and (6) for winglet fitting. Columns (1)-(3) show the results for a sample that excludes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Eastern Europe). Columns (4)-(6) shows the results when the sample only includes routes from the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Western Europe). Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

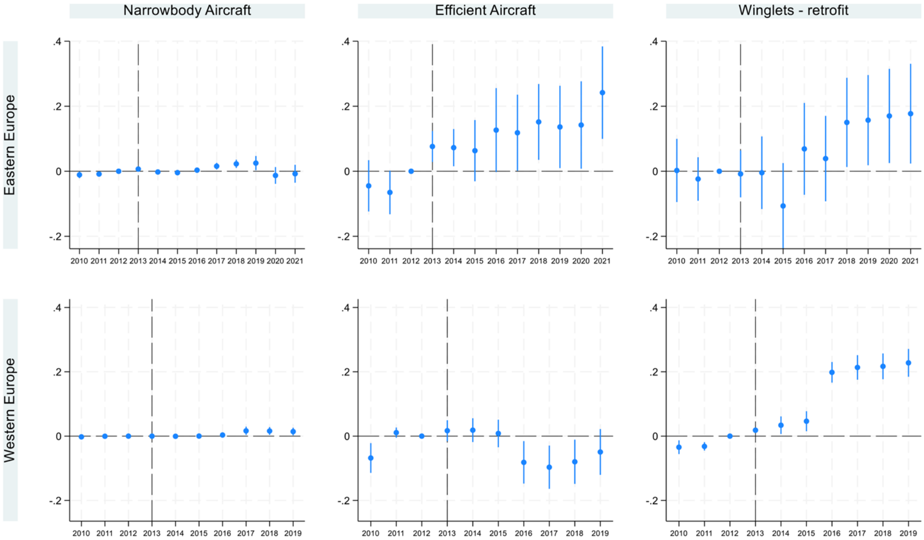

Fig. 9 Event-study estimates for aircraft choice (Eastern vs. Western Europe). Notes: This figure plots the results from an event-study analysis of the differences in the type of aircraft used. The top panels show the annual effects of the EU ETS on the use of different type of aircraft (narrowbody aircraft, per year-distance fuel-efficient aircraft, aircraft with winglet installation) when the sample excludes the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Eastern Europe). The bottom panels show the annual effects on the use of different type of aircraft when the sample only includes routes from the 15 pre-2004 EU member countries, Iceland, Norway, and Switzerland (Western Europe). 2012 is used as a reference year. The confidence interval is set at 95% and standard errors are clustered at the route level

for either Eastern or Western Europe. Only some significant point estimates arise after 2015 and before the pandemic outbreak, but the size of the effect is below 2%.

Altogether, these results show that the EU ETS is responsible for an improvement in emissions driven by actual technology change. In this regard, the size effect on emission intensity seems, in this case, to be driven, firstly, by the timing of the technology adoption (in the main, a few years after policy implementation and potentially accompanying the higher carbon prices). Secondly, technological change is heterogenous in Europe: Western Europe favors retrofit investments, while Eastern Europe favors aircraft updates. In the following section, we perform a mediator analysis to disentangle the extent to which these technological changes explain the changes in emission intensity.

6 Mechanisms

Our previous analysis provides evidence that the EU ETS has led to lower emissions per seat-km. We also find that it incentivizes the use of more fuel-efficient planes and a higher rate of winglet installments (retrofit). To examine the link between these outcomes, we implement a mediator analysis (Baron and Kenny 1986; Pearl 2012) to assess the extent to which the effect of the EU ETS on emission intensity is mediated by aircraft choices. We use a stepwise method in which we estimate the following equations at the route-airline level, controlling for the same covariates as in previous regressions:

$$\log\left(emission\_intensity\right)_{it} = \alpha\ + \beta\, ETS_{it} + \lambda\, X_{it} + \gamma_i + \eta_t + \epsilon_{it} \tag{5}$$

$$Share\_aircraft\_type_{it} = \alpha\ + \rho\,_k ETS_{it} + \lambda\, X_{it} + \gamma_i + \eta_t + \epsilon_{it} \tag{6}$$

$$\begin{aligned}\log\left(emission\_intensity\right)_{it} = \alpha \\ + \beta\,'\, ETS_{it} + \nu\,_k Share\_aircraft\_type_{it} \\ + \lambda\, X_{it} + \gamma_i + \eta_t + \epsilon_{it}\end{aligned} \tag{7}$$

where β is the total effect of the EU ETS on emission intensity (from Eq. 5), previously estimated in Table 2. ρ_k is the effect of the EU ETS on the share of aircraft type *k* at the airline-route level (in Eq. 6). Finally, *β'* is the effect of the EU ETS after controlling for the share of aircraft type *k* (in Eq. 7).

For mediation to hold, three conditions must be satisfied: (i) ETS must affect emissions intensity ($\beta \neq 0$), as confirmed in Table 2; (ii) ETS must influence the share of aircraft type ($\rho \neq 0$), supported by regressions in Table 7 and Table A2 (using different data at the airline-route-year level); and (iii) the aircraft type share must affect emissions intensity ($\nu \neq 0$). If all conditions are met, then the effect of the ETS variable on emissions intensity must be lower in Eq. 7, i.e., $\beta' < \beta$. If β' is not statistically significant, it suggests full mediation. Otherwise, other mechanisms also contribute, indicating partial mediation.

Some caveats apply. First, the *ETS* and *share_aircraft-type* are correlated in Eq. 7. Second, the effect of the narrowbody share variable may be captured by the aircraft seats variable, which is included as a control in previous regressions to disentangle technological choices from the size of the aircraft: replacing a regional jet with a narrowbody aircraft may reflect a low-carbon shift, but seat differences also affect emissions per seat-km. Third, fuel-efficient aircraft are often narrowbodies with winglets, making it difficult to identify the specific influence of each aircraft type.

Table 8 reports the results of this mediator analysis. We find that fuel-efficient aircraft and winglet retrofits act as mediating variables: they affect emissions intensity and attenuate the estimated effect of the ETS. In contrast, while the use of narrowbody aircraft increases in response to the EU ETS, it does not significantly mediate the effect. That said, all aircraft classified as fuel-efficient (conditional on distance and year) are also narrowbody.

Therefore, we conclude that aircraft choices, particularly fleet renewal and retrofit adoption, are important mechanisms through which the EU ETS reduces emissions per seat-km. However, as the ETS variable remains significant after controlling for the share of aircraft type, we do not find evidence of perfect mediation.

As we have mentioned above, we use detailed supply data that allow us to consider a measure of CO_2 emissions per seat-km. This measure is based on the type of aircraft used, which affects emissions through their size and fuel efficiency. Furthermore, it is determined by the seat configuration of the aircraft and the number of kilometers flown that will affect the seats-km measure. Therefore, the lack of perfect mediation by aircraft choices may be partly explained by limitations in the variable used to capture the adoption of the most efficient aircraft, which is a binary variable that does not distinguish among the remaining, less efficient aircraft. On the other hand, other factors that may have an influence on these results include the relative intensity in the use of the least efficient aircraft, the seat configuration for each aircraft model, and the composition of routes in terms of distance.

To explore the role of these potential mechanisms in driving the effect of the ETS on emissions per seat-km, Table 9 presents the results from additional regressions that use dif-

Table 8 EU ETS and emission intensity (mediator analysis)

Variables	(1)	(2)	(3)	(4)
ETS	-0.0212***	-0.0210***	-0.0153***	-0.0173***
	(0.00318)	(0.00317)	(0.00299)	(0.00316)
Share narrowbody		-0.00383		
		(0.00994)		
Share per-distance year fuel efficient			-0.123***	
			(0.00617)	
Share winglet				-0.0534***
				(0.00229)
Observations	158,891	158,891	158,891	158,891
R-squared	0.973	0.973	0.975	0.973
Route-airline FE	YES	YES	YES	YES
Year FE	YES	YES	YES	YES
Sample	All	All	All	All
Period	2010–2021	2010–2021	2010–2021	2010–2021
Clusters	Route	Route	Route	Route
Controls	All	All	All	All

Note: This table shows the results of the mediator analysis. Column (1) shows the results of Eq. (5), while columns (2) to (4) show the results of Eq. (7) distinguishing between the different types of aircraft considered

Table 9 EU ETS and emission intensity (other mechanisms)

Variables	(1) Flights	(2) Distance	(3) Aircraft_seats	(4) Aircraft vintage
ETS	-0.163***	-0.0321	-0.00651***	0.591***
	(0.0145)	(0.0268)	(0.00236)	(0.0644)
Observations	159,569	159,569	530,162	530,162
R-squared	0.945	0.508	0.995	0.497
Year FE	YES	YES	YES	NO
Route FE	YES	NO	YES	YES
Airline FE	YES	YES	YES	YES
Aircraft FE	NO	NO	YES	NO
Sample	All	All	All	All
Period	2010–2021	2010–2021	2010–2021	2010–2021
Clusters	Route	Route	Route	Route
Controls	All	All	All	All

Note: This table shows difference-in-differences coefficients using different outcome variable; total number of flights (column 1), route distance in kms (column 2), aircraft seats per specific aircraft model (column 3) and the first year of the introduction of the aircraft used (column 4). Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

ferent dependent variables, retain the same control variables as in previous regressions and apply various sets of fixed effects. Regressions in columns (1) and (2) use the dataset at the route-airline-year level, while regressions in columns (3) and (4) use the dataset at the aircraft-route-airline year level. In column (1), we consider the total number of flights as dependent variable to capture the effect of the intensity in the use of the aircraft fleet. In column (2), we consider the route distance as dependent variable considering the negative relationship between distance and emissions per seat-km. Here, we do not add route-fixed effects that would capture the effect related to distance. In column (3), we consider the aircraft size as dependent variable, considering the same fixed effects as in the baseline regressions but also aircraft fixed effects. This allows us to isolate the effect of seat configuration for each specific aircraft model. In column (4), we consider the first year of the introduction of the aircraft used in each aircraft-route-airline-year combination considering that newer aircraft are more fuel-efficient. Here, we do not add year fixed effects that capture all the effect given the general trend across airlines to progressively retire old aircraft and/or buy newer ones.

We find a strong and statistically negative impact of the EU ETS variable on the number of flights, which are about 17% lower on treated routes as compared to the counterfactual. While this finding is consistent with previous studies that report a causal negative impact of the EU ETS on the number of flights, even in a context of low carbon prices (Fageda and Teixidó 2022; Kang et al. 2022), the magnitude of the effect we identify does not necessarily constitute evidence of a causal impact of the EU ETS on flight numbers, which lies beyond the scope of this paper. Our aim here is instead to identify the context within which airlines made decisions during the period under consideration regarding the use of their aircraft fleet. Keeping this in mind, one potential driver of this result is a lower relative use of the least efficient aircraft on treated routes.

Furthermore, we find a decrease in both the number of kilometers flown and the number of seats per aircraft model on treated routes, although the effect for distance is not statistically significant. This leads to fewer seat-km on treated routes, given the type of aircraft used, which could lead to an underestimation of the ETS impact on emissions intensity related to aircraft choices. Finally, we find evidence of greater use of newer aircraft on treated routes, supporting the view that the ETS encourages the deployment of newer, and therefore more efficient, aircraft. Overall, we can therefore conclude that the main mechanism explaining the effect of the ETS on emissions intensity is the more intensive use of efficient aircraft.

This analysis is based on aircraft usage at the route-airline level, which does not allow us to assess the potential role of selling older aircraft to airlines operating in other markets. In the sample analyzed, we observe a general trend toward reduced emissions intensity on both treated and control routes, with the decline being observed somewhat faster on treated routes. The largest effect is found in the Eastern sample, where treated routes consistently use less efficient aircraft than control routes throughout the entire period examined. This suggests that the effect in the sample is unlikely to be driven by the sale of prematurely retired aircraft to airlines operating on control routes. However, we cannot identify whether aircraft are being sold by EEA airlines to non-EEA airlines from low-income countries not included in our analysis. In any case, it is not clear whether such potential spillover effects would be positive or negative, as non-EEA airlines would not necessarily be acquiring better-performing older aircraft in the absence of the EU ETS.

Another mechanism that could explain the effect of the ETS on emissions intensity is the redeployment of more efficient aircraft from control to treated routes. This might be particularly relevant for control routes connecting an EEA airport with a non-EEA airport –for example, relocating the most efficient planes from the Frankfurt-Istanbul route to the Frankfurt-Barcelona route. This is plausible because airlines may simultaneously operate on treated routes and on routes linking an EEA airport with a non-EEA airport. However, this does not appear to be the case: when we limit the control group to routes connecting two non-EEA airports (i.e.; excluding routes like the Frankfurt-Istanbul), the estimated effect is even larger. A caveat worth mentioning is that airlines operating routes between two EEA airports are usually different from those serving routes between two non-EEA airports. Nonetheless, we cannot entirely rule out some redeployment of EEA airlines' aircraft outside the EEA.

Finally, our analysis reveals that a higher relative use of more efficient aircraft is a key driver of the effect of the EU ETS on emissions intensity. This pattern is consistent with a faster fleet renewal as also found by De Jong (2022). Descriptive evidence shows improvements in emissions intensity and a greater use of more efficient aircraft on both treated and control routes, which is consistent with the influence of broader motivations like financial or strategic considerations. However, carbon reduction efforts spurred by the EU ETS may help explain why this general trend toward increased use of the most efficient aircraft is somewhat faster on routes affected by the policy.

7 Discussion and Conclusions

Carbon markets are designed to reduce emissions in a cost-effective manner, a process that includes the spurring of low-carbon innovation and, in the long term, technology diffusion. To date, empirical studies on carbon markets have, indeed, shown that they spur low-carbon innovation, with regulated firms increasing their low-carbon patenting and R&D spending. However, evidence from these greater innovation efforts being transferred into the greater adoption of these technologies, which is ultimately what will reduce emissions, is rather ambiguous. Without the diffusion of low-carbon technologies, carbon markets are unable to realize their full potential in terms of cost-effectiveness in reducing emissions. Here, we report novel evidence of diffusion of low-carbon technology in response to carbon pricing, isolating the role played by two mechanisms: an earlier adoption of more fuel-efficient aircraft and the retrofitting of planes in use (i.e., winglet installment). The retrofitting of winglets is the strongest adoption practice identified, especially after 2015, that is, after carbon emissions started being priced in the sector and mainly located in flights operating in Western Europe. Likewise, the replacement of aircraft for more efficient models, although slow, presents a higher substitution rate on ETS routes than on those of the counterfactual and proves to be the main causal mechanism on Eastern European routes.

The impact of the ETS that we find is notable in a context in which carbon prices have been low for most of the sampled period, in which the general trend of the sector is towards a reduction in emission intensities, and the technological options available are limited. In this regard, we show that high carbon prices can have strong positive effects on the adoption of low-carbon technologies. While the specific dynamics of technology adoption may vary across sectors, the broader patterns observed in aviation offer valuable insights into how carbon pricing can drive low-carbon technology diffusion in other hard-to-abate sectors, which remain critical to meeting global climate targets.

These results, however, need to be considered in context. From a policy perspective, the emission reductions achieved to date fall short of climate targets in this sector: emissions in the aviation sector have yet to stop increasing. The EU ETS has only moderated emission growth by reducing supply and, as we show in this paper, by spurring the adoption of low-carbon technology. The bottom line remains, as economic theory is quick to remind us, that carbon pricing tends to work by promoting only those technology changes whose implicit abatement costs are below the effective carbon price. In this regard, our results suggest differences in abatement costs between higher income Western economies and lower income Eastern economies determine the availability of technological choices. Moreover, our results are particularly timely in a context in which other measures with much higher CO_2 abatement costs, such as blending mandates for biofuels, are gaining relevance.[21] Until biofuels become economically viable for airlines, i.e. cost-competitive with the carbon price, the carbon market is key to continuing to provide cost-effective incentives for adopting abatement technologies.

Higher carbon prices may well further encourage the adoption of low-carbon technology. Here, the marked rise in the fitting winglets coinciding with the approval of the MSR (and an increase in carbon prices) is consistent with this belief. However, as with other climate policies, this also involves an economic cost that may have a knock-on effect on consumer prices. In the particular case of the aviation sector, however, this raises fewer equity concerns, since air travel is highly correlated with income (Cass and Lucas 2022; Gössling and Humpe 2020; O'Garra and Fouquet 2022).

8 Appendix

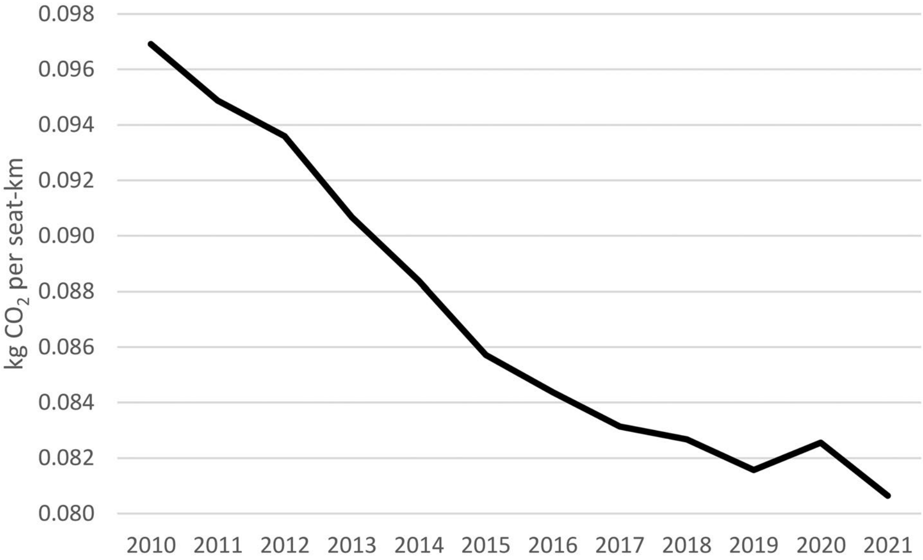

Fig. A1 Emission intensity in the European aviation market. Notes: This figure plots the evolution in average emission intensity, measured in kg of CO_2 per seat-km, in the European market including treated and control routes

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[21] Minimum shares of biofuels have been imposed by the European Commission (European Commission 2021). Such minimum shares will progressively increase, starting with 2% in 2025, 5% in 2030, 20% in 2035, 32% in 2040, 38% in 2045 and finally 63% in 2050.

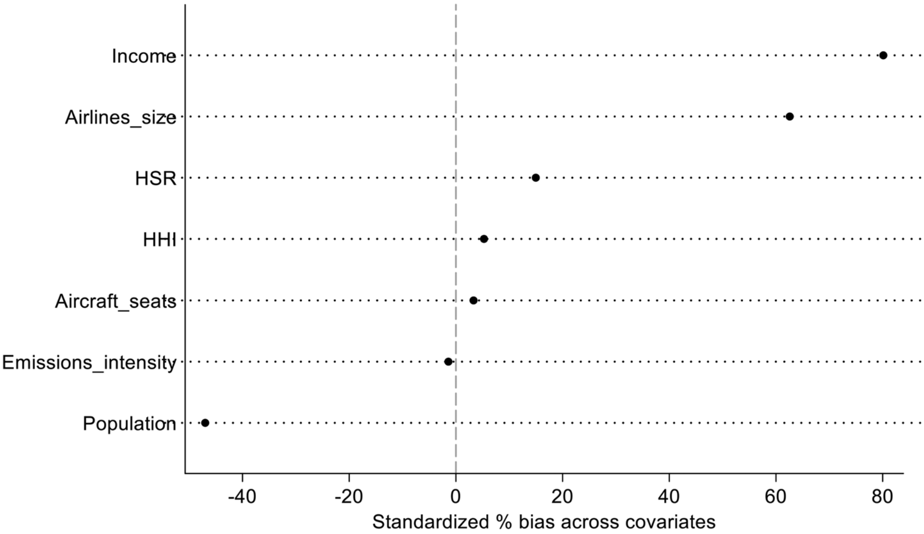

Fig. A2 Standardized mean difference for covariates in treated and control routes (pre-treatment period). *Note*: This graph plots the standardized differences in % between treated and control routes before 2013

Fig. A3 Carbon prices in the EU ETS (2010–2021).
Source: ICAP

Table A1 EU ETS and emission intensity (emissions per seat-km)

Variables	(1)	(2)	(3)	(4)	(5)	(6)
ETS	-0.0185***	-0.0154***	-0.0212***	-0.0203***	-0.0229***	-0.0418***
	(0.00291)	(0.00446)	(0.00318)	(0.00314)	(0.00355)	(0.00505)
HHI	0.0111***		0.0121**	0.0131**	0.0131**	0.0108*
	(0.00270)		(0.00560)	(0.00631)	(0.00555)	(0.00622)
HSR	0.0960**		0.0622***	0.0546***	0.0639***	0.0617***
	(0.0422)		(0.0122)	(0.0151)	(0.0130)	(0.0120)
Aircraft_seats	-0.327***		-0.345***	-0.381***	-0.342***	-0.347***
	(0.0115)		(0.0144)	(0.0164)	(0.0145)	(0.0157)
Income	-0.0210**		-0.00994	-0.0133	-0.0111	-0.00906
	(0.00888)		(0.0149)	(0.0155)	(0.0151)	(0.0155)
Population	-0.125***		-0.157***	-0.0610**	-0.158***	-0.226***
	(0.0199)		(0.0309)	(0.0310)	(0.0319)	(0.0347)
Airline_size	0.00746***		0.0122***	0.00786***	0.0127***	0.00944***
	(0.00170)		(0.00236)	(0.00278)	(0.00245)	(0.00256)
Observations	158,891	158,976	158,891	131,807	146,765	136,238
R-squared	0.975	0.947	0.973	0.977	0.973	0.974
Route-airline FE	YES	YES	YES	YES	YES	YES
Year FE	YES	YES	YES	YES	YES	YES
Sample	All	All	All	All	All	Sutva
Period	2010–2021	2010–2021	2010–2021	2010–2019	2010–2021	2010–2021
Year 2012	YES	YES	YES	YES	NO	YES
Clusters	Route	Route	Route	Route	Route	Route
Weights	NONE	Flights	Flights	Flights	Flights	Flights

Note: This table shows difference-in-differences coefficients using as our outcome variable a measure of flight emission intensities (emissions per seat-km). Column (1) shows the results without applying weights, while the rest of columns show the results with the sample weighted by the number of flights. Colum (3) shows a regression without controls to check their influence on the results. To check if the COVID-19 pandemic may distort our results, column (4) considers the period 2010–2019. To see if the effect could be affected by the uncertainty in 2012, column (5) shows results when excluding this year. In column (6), the control group only includes those routes flying within non-EEA countries to check whether the potential violation of SUTVA assumption could distort our results. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

Table A2 EU ETS and share aircraft type

Variables	(1)	(2)	(3)
	share_narrowbody	Share per-distance year fuel efficient	share_ winglet
ETS	0.0391***	0.0362***	0.0540***
	(0.00599)	(0.00892)	(0.00699)
Observations	158,891	158,891	158,891
R-squared	0.932	0.895	0.736
Route-airline FE	YES	YES	YES
Year FE	YES	YES	YES
Sample	All	All	All
Period	2010–2021	2010–2021	2010–2021
Clusters	Route	Route	Route
Controls	All	All	All

Note: This table shows difference-in-differences coefficients using the share by aircraft type as outcome variable. Robust standard errors (in parentheses). *** $p<0.01$, ** $p<0.05$, * $p<0.1$

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Author Contributions All authors contributed to the study conception and design. Xavier Fageda: Writing, Methodology, Investigation, Formal analysis, Data curation, Conceptualization, Visualization. Jordi J. Teixidó: Writing, Methodology, Investigation, Formal analysis, Data curation, Conceptualization, Visualization.

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Declarations

Conflict of interest The authors declare no relevant conflict of interest, and have provided details here of all financial support received for this research.

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