Beyond Zero-Emission Labels:

A Region-Specific Use-Phase Assessment of Electric Vehicles in China and the United States

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Abstract

Electric vehicle subsidies in China and the United States are usually designed as nationally or federally uniform instruments, implicitly assuming that the environmental benefit of replacing an internal combustion engine vehicle (ICEV) with a battery electric vehicle (BEV) is constant across regions. This study tests that assumption by estimating the per-kilometer net emission impact of BEV-for-ICEV substitution across 31 Chinese provinces and 51 U.S. states, using a harmonized three-phase framework that combines Fisher-Pry adoption forecasting, multi-pollutant use-phase emission accounting (CO2, NOx, PM2.5, PM10), and multi-criteria policy scoring. The environmental return is found to be highly heterogeneous, varying 2 to 6 fold across regions with local grid composition and fleet structure. BEV substitution reduces CO_2 in every region but diverges sharply on local pollutants: it worsens NOx in 16 of 31 Chinese provinces yet reduces NOx in all 51 U.S. states, a contrast traceable to the substantially older U.S. vehicle fleet and its higher tailpipe baseline. Sales-weighted data further show near mass-parity between BEVs and ICEVs in China but a 15% heavier BEV fleet in the United States, producing negative net PM2.5 effects in five coal-dependent states. These results imply that uniform subsidies are inefficient as environmental policy. The paper proposes a differentiated, tier-based incentive framework aligning subsidy intensity with measured environmental benefit, market scale, and air-quality baseline, and discusses financing mechanisms suited to each country's institutional context.

1. Introduction

Transportation system decarbonization has become a major instrument for governments confronting climate change. The transportation sector contributes roughly 23% of the world's energy-related CO_2 emissions (Mead, 2021), and vehicle electrification is widely promoted as a primary pathway to reduce emissions. China and the United States, the world's two largest automotive markets and top carbon emitters, have each committed substantial fiscal resources to accelerate the transition from internal combustion engine vehicles (ICEVs) to new energy vehicles (NEVs), such as battery electric vehicles (BEVs) and plug-In hybrid electric vehicle (PHEV). Yet consumer enthusiasm for electric vehicles in both countries has often been coupled with the misconception that EVs are “zero-emission”. This perspective focuses narrowly on the absence of ICEV's tailpipe emissions while overlooking NEV's upstream and indirect environmental impacts.

This study aims to build a research demonstrating that the environmental performance of NEVs, especially BEVs, is condition-dependent rather than uniform. Life-cycle assessments generally conclude that NEVs deliver net carbon reductions relative to ICEVs despite higher manufacturing emissions (Negri & Bieker, 2025), but the magnitude of the use-phase advantage hinges on where and how the vehicle is operated. Regional and temporal variation in the upstream emissions of electricity generation can substantially alter, or even reverse, the climate advantage of electrification (Singh et al., 2024; Holland et al., 2016). In China, provinces such as

Sichuan and Yunnan are rich in hydropower, while Inner Mongolia, Hebei, and much of northern China remain heavily coal-dependent. In the United States, the contrast between hydro-rich Washington and coal-dominant West Virginia is equally stark. A parallel concern is the rising significance of non-exhaust emissions, particularly tire wear particles (TWP), as tailpipe pollutants decline. Because TWP emissions scale with vehicle mass, and because NEVs carry heavy battery packs, the relative curb mass of the two vehicle types directly shapes their particulate footprint (Liu et al., 2021; Giechaskiel et al., 2024).

These dependencies imply that the net effect of replacing an ICEV with a NEV is fundamentally a local quantity, varying with the carbon and pollutant intensity of the regional grid, the age and emission-standard composition of the displaced fleet, ambient temperature, and the mass of the vehicles. Capturing this requires moving beyond a single national average and a single pollutant. This study therefore evaluates NEV-for-ICEV substitution across four pollutant dimensions simultaneously: CO_2, alongside the local air pollutants NOx and particulate matter (PM2.5 and PM10) that dominate the public-health burden of road transport. On the ICEV side, tailpipe emission factors are differentiated by the emission-standard vintage composition of each region's in-use fleet rather than assumed uniform. On the NEV side, the analysis accounts for not only CO_2, but NOx, SO_2, and PM emitted at power plants during charging. Both vehicle types receive temperature corrections reflecting local climate, and the representative mass and energy parameters of each are derived from 2025 sales-weighted vehicle data.

This sales-weighted approach surfaces a result worth stating at the outset. In the 2025 Chinese market, sales-weighted NEV and ICEV curb masses are nearly identical, at 1,581 kg versus 1,603 kg. In the United States, by contrast, NEVs are roughly 15% heavier, at 2,106 kg versus 1,833 kg, a divergence with direct implications for the tire-wear comparison and one that varies sharply between the two markets. The study further introduces a temporal dimension through Fisher-Pry diffusion forecasting of adoption through 2030, and it conducts the entire evaluation in parallel for 31 Chinese provinces and 51 U.S. states.

The central argument is that the environmental returns to NEV substitution are highly heterogeneous across subnational regions, driven primarily by local grid composition and existing fleet structure, and that the uniform-subsidy designs in China and the less resolute subsidy designs in the United States therefore leave substantial efficiency gains unrealized. The objective is not merely to document this heterogeneity but to translate it into an actionable, differentiated policy framework that aligns subsidy intensity with measured environmental benefit, market scale, and air-quality urgency.

Three research questions structure the analysis:

- Question 1 asks how the per-kilometer net environmental impact of

NEV-for-ICEV substitution, measured across CO_2, NOx, PM2.5, and PM10, varies across Chinese provinces and U.S. states.

- Question 2 asks which structural factors drive this variation, considering grid composition, fleet emission-standard distribution, climate, and vehicle mass, and how these factors differ between the two countries.
- Question 3 asks how these findings can inform a differentiated subnational NEV incentive framework that maximizes environmental returns.

The remainder of this paper proceeds as follows. Section 2 traces the historical evolution of EV incentive policies in China and the United States. Section 3 details data sources and the three-phase methodology. Section 4 presents results. Section 5 develops the policy analysis and the proposed differentiated framework. Section 6 discusses limitations, and Section 7 concludes.

2. Policy Context of NEV Subsidies in China and the United States

2.1 A Shared Goal Under Different Systems

China and the United States are the world's two largest automotive markets and the two most consequential testing grounds for electric vehicle incentive policy. According to data released by the China Association of Automobile Manufacturers, in 2025, China's NEV (including BEV) sales reached 16.49 million units, representing 47.9% of all new passenger vehicle sales in the country. The United States, in parallel, recorded approximately 1.3 million BEV sales in 2024, representing a market share of roughly 8 to 10 percent.

Meanwhile, both countries have deployed substantial fiscal resources to drive this transition. According to clearance audit disclosures by the Ministry of Industry and Information Technology's Equipment Industry Development Center, central government subsidies for NEV deployment in China exceeded RMB 100 billion over the program's duration from 2009 to 2022. In the United States, the Inflation Reduction Act of 2022 established consumer tax credits of up to USD 7,500 per qualifying new electric vehicle, with projected ten-year fiscal exposure revised upward in subsequent Congressional Budget Office assessments.

Despite the considerable differences in institutional structure, policy instruments, and implementation pace between the two countries, their national subsidy architectures share a critical common feature: both are designed as uniform instruments, applying the same incentive level regardless of where in the country the vehicle is purchased and operated. This chapter traces the historical evolution of NEV incentive policy in each country, documents the key turning points, and establishes why this uniformity assumption on the national level has persisted even as evidence of geographic heterogeneity in environmental outcomes has grown.

2.2 China: Three Phases from Public Demonstration to Market Competition

China's consumer-facing NEV incentive architecture has rested on two parallel fiscal instruments. The first is a direct per-vehicle purchase subsidy, administered through annual central government appropriations. The second is an exemption from China's vehicle purchase tax, which is otherwise levied at a standard rate of 10% of the pre-tax sale price. The two instruments operate through different mechanisms: the purchase subsidy reduces the headline price visible to the consumer at the point of sale, while the purchase tax exemption reduces the fiscal burden imposed separately at registration. The vehicle purchase tax exemption, introduced in September 2014 and subsequently renewed in 2017, 2020, and 2022, has followed a more continuous path. However, the purchase subsidy underwent the more sharply defined policy trajectory, the discussion that follows is organized primarily around its evolution.

2.2.1 The Demonstration Phase (2009 to 2015)

China's NEV support policy originated in early 2009, when the Ministry of Finance, Ministry of Science and Technology, National Development and Reform Commission, and Ministry of Industry and Information Technology jointly launched the Ten Cities, Thousand Vehicles program. The program aimed to deploy 1,000 NEVs in each of ten cities per year over three years, concentrating initially on public bus fleets, municipal service vehicles, and taxis rather than private passenger vehicles. The first group included thirteen cities, and central government subsidies for a single BEV reached as high as RMB 60,000 per unit in this early period, with local government co-subsidies bringing total support in some cases close to RMB 100,000 per vehicle (Ministry of Finance et al., 2009).

In 2012, the State Council issued the Energy-Saving and New Energy Vehicle Industry Development Plan (2012 to 2020), which formally extended the purchase subsidy framework to the private market and set explicit targets for cumulative NEV sales and production capacity by 2020 (State Council of the People's Republic of China, 2012). The underlying policy logic in this phase was characteristic of supply-side industrial cultivation: use concentrated public-sector procurement to build out the NEV supply chain, reduce unit costs, and accumulate technical capability before shifting to a consumer market orientation.

2.2.2 The Tapering Phase (2016 to 2022)

As the NEV market expanded, the sustainability of direct fiscal subsidies came under pressure. Beginning in 2016, the central government implemented an annual phasedown of per-vehicle subsidy amounts, while simultaneously raising eligibility thresholds. Vehicles, regardless BEVs or PHEVs, below minimum drive range on fully electric mode were progressively excluded from subsidy eligibility, and requirements for battery energy density and vehicle energy consumption were tightened and made binding parameters for the subsidy amount received (Ministry of Finance et al., 2021). By 2022, the final year of the program, the standard subsidy had declined to RMB 12,600 per pure electric vehicle and RMB 4,800 per plug-in hybrid, from the original ceiling of RMB 60,000.

The most consequential institutional innovation of this phase was the Dual Credit Policy, formally titled the Parallel Administration of Corporate Average Fuel Consumption and New Energy Vehicle Credits for Passenger Vehicle Enterprises. The policy was published by the Ministry of Industry and Information Technology in September 2017 and took effect on April 1, 2018, with NEV credit compliance requirements beginning in 2019 (Ministry of Industry and Information Technology et al., 2017). The Dual Credit system simultaneously governed two types of credits: corporate average fuel consumption credits, known as CAFC credits, and new energy vehicle credits, known as NEV credits. NEV positive credits could be freely traded on an exchange platform and used to offset both NEV deficits and CAFC deficits. The policy introduced a market-based compliance mechanism in which automakers with excess combustion engine production effectively transferred value to those generating NEV credits, creating a structural incentive for electrification that operated independently of direct fiscal transfers. This design was, at the time of its introduction, the first instance globally of parallel management of fleet fuel economy and NEV production mandates within a unified regulatory framework.

2.2.3 The Post-Subsidy Phase (2023 to the Present)

On December 31, 2022, the central government's purchase subsidy program formally terminated, as previously announced in a joint notice by the Ministry of Finance, Ministry of Industry and Information Technology, and other agencies at the end of 2021 (Ministry of Finance et al., 2021). Vehicles registered after that date were no longer eligible for the direct per-vehicle central subsidies that had defined Chinese NEV policy for thirteen years.

Fiscal support did not disappear entirely, but shifted toward tax-based instruments. The purchase tax exemption for qualifying NEVs, which had been in effect since 2014 and extended multiple times, was renewed and extended through the end of 2027 (Ministry of Finance et al., 2023). In parallel, provincial and municipal governments maintained a patchwork of local incentives through 2023 to 2025, creating considerable variation in the effective subsidy available to buyers across regions.

This fragmentation was addressed at the start of 2026, when the Ministry of Commerce and seven co-issuing agencies published the 2026 Automobile Trade-In Subsidy Implementation Rules. The policy established a nationally uniform subsidy structure for the trade-in program, replacing the prior fixed-amount per-vehicle design with a price-proportional system: consumers scrapping an eligible older vehicle and purchasing a qualifying NEV receive a subsidy equal to 12% of the new vehicle's sale price, capped at RMB 20,000; consumers substituting through a sale-and-purchase transaction rather than scrapping receive 8% of the new vehicle price, capped at RMB 15,000 (Ministry of Commerce et al., 2025). It bears emphasis that this trade-in subsidy is a targeted demand stimulus for vehicle replacement, not a universal purchase subsidy available to all NEV buyers. Its significance for this study is that it represents the Chinese government's move toward nationally standardized incentive parameters once again after a period of provincial fragmentation and leaves future efficiency gains unrealized.

This trajectory, from generous universal purchase subsidies through tapering, to post-subsidy tax instruments and targeted trade-in incentives, reflects a deliberate shift in policy logic: direct fiscal incentives served to incubate a market that the government judged ready to sustain itself through market forces, regulatory mandates, and residual tax support. The NEV industry's sustained growth through and beyond the subsidy exit, with sales reaching 16.49 million units in 2025, has been cited as evidence that this sequencing succeeded (China Association of Automobile Manufacturers, 2026). What the sequencing did not address, however, is the geographic dimension of environmental returns. The national subsidy was designed to maximize aggregate market volume, not to allocate fiscal resources toward the regions where NEV substitution generates the largest per-unit environmental benefit.

2.3 The United States: Federal Tax Credits, State-Level Divergence, and Policy Reversal

2.3.1 Origins and the 200,000-Vehicle Phaseout Mechanism (2008 to 2017)

The federal electric vehicle tax credit was established by the Energy Improvement and Extension Act of 2008 and reinforced by the American Recovery and Reinvestment Act of 2009. It provided a maximum credit of USD 7,500 per qualifying plug-in electric vehicle, structured as a nonrefundable income tax reduction rather than a direct payment, which directed its benefits disproportionately toward higher-income buyers with sufficient federal tax liability to utilize the full amount (IRS, Notice 2009-89).

A defining architectural feature was a manufacturer-level phaseout: once a single automaker had sold a cumulative 200,000 qualifying electric vehicles in the United States, the credit for that manufacturer's vehicles stepped down over four quarters and then terminated. This created a structural paradox in which the most commercially successful EV manufacturers were the first to lose federal support. Tesla reached the threshold in mid-2018 and lost eligibility entirely by end of 2019; General Motors followed, with full phaseout completing by April 2020 (IRS, 2022). Alongside the federal credit, California's Zero Emission Vehicle mandate has operated as a parallel and more durable instrument. Under authority granted by the federal Clean Air Act, California has required automakers to sell increasing proportions of zero-emission vehicles since 1990, with other states permitted to adopt California's standards in lieu of federal ones.

2.3.2 The IRA Restructuring (2022 to 2024)

The Inflation Reduction Act of 2022 restructured the federal credit substantially. It eliminated the per-manufacturer 200,000-vehicle cap and established a maximum credit of USD 7,500 for new qualifying BEVs and USD 4,000 for used qualifying BEVs (Congressional Research Service, 2024). In exchange, it introduced significantly more stringent eligibility conditions: final vehicle assembly must occur in North America, and the USD 7,500 credit is split into two USD 3,750 components each contingent on battery mineral and component sourcing thresholds, with additional buyer income limits and vehicle price caps. These conditions were designed to link EV incentives to domestic manufacturing and battery supply chain localization, and their effect was to sharply narrow the qualifying vehicle list, from 14 models in 2023 to 4 at the start of 2024 (Car and Driver, 2024). On the state regulatory side, the California Air Resources Board adopted Advanced Clean Cars II in August 2022, requiring 35% ZEV sales by model year 2026, rising to 100% by 2035, with multiple states subsequently adopting the standard under Section 177 of the Clean Air Act (CARB, 2022).

2.3.3 Federal Withdrawal and State Divergence (2025 to the Present)

The change of administration in January 2025 produced an abrupt policy reversal. In June 2025, the Trump administration signed three Congressional Review Act resolutions repealing California's Clean Air Act waivers for Advanced Clean Cars II and related rules, and California responded by directing its Air Resources Board to develop a replacement framework, Advanced Clean Cars III (Climate Policy Dashboard, 2025). Separately, legislation enacted as Public Law 119-21 terminated all three IRA electric vehicle tax credits for vehicles acquired after September 30, 2025, with the Congressional Research Service projecting the repeal will reduce federal deficits by approximately USD 190 billion over ten years (Congressional Research Service, 2025).

The result is a period of federal disengagement and state-level fragmentation. States that had aligned with California's ZEV standards face regulatory uncertainty; those without comparable state-level incentives now confront the full withdrawal of federal support. The United States has, in the span of three years, moved from the most ambitious restructuring of its NEV incentive framework in the program's history to the program's near-complete elimination at the federal level.

3. Data and Methodology

3.1 Analytical Scope and Boundary Conditions

Unit of comparison: the substitution analysis compares BEVs against ICEVs. PHEVs are only included in the Phase 1 diffusion forecast (under the official NEV definition) but excluded from the Phase 2 environmental assessment. The exclusion is conceptual, since PHEV's emissions depend on its electric-versus-combustion driving share, which varies with individual charging behavior and admits no stable fleet-level utility factor for clean comparison against an ICEV baseline. BEVs carry no such ambiguity, making the BEV-versus-ICEV pairing the analytically tractable unit.

Lifecycle boundary: the analysis covers the use phase, including tailpipe emissions, upstream electricity emissions from charging, and tire wear. Manufacturing and end-of-life phases are excluded, for three reasons:

- Dominance: the use phase drives the large majority of ICEV lifecycle emissions. DOE GREET attributes roughly 73% of ICEV lifecycle GHG to fuel combustion (DOE, 2024); LCA studies place the operational share at 73-83% over a 200,000 km lifetime, versus 17-20% for manufacturing and ~5% for end-of-life (Tang et al., 2022). This is where policy has the greatest marginal leverage.
- Geographic orthogonality: manufacturing emissions are tied to where a vehicle is produced, not where it is operated. Since this study examines spatial variation in environmental returns driven by local grid and fleet structure, including production emissions would inject a cross-border attribution problem irrelevant to the policy question.
- Methodological comparability: end-of-life battery accounting lacks consensus; assumptions on recycling recovery, second-life use, and material credits vary widely across studies, adding noise rather than precision.

Pollutant scope: four dimensions, including CO_2, and NOx, PM2.5, PM10. SO_2 from generation is tracked as a supplementary variable but excluded from composite scoring given its lower direct health relevance in these contexts.

3.2 The Three-Phase Framework

- Phase 1: forecasts BEV/NEV adoption through 2030 via a Fisher-Pry diffusion model.
- Phase 2: quantifies the per-kilometer net emission impact of BEV substitution across four pollutants, in five steps.
- Phase 3: combines Phase 2 environmental results, Phase 1 market scale, and air quality baselines into a composite score and policy tier classification.

3.3 Phase 1: Fisher-Pry Diffusion Forecasting

NEV penetration in each region is modeled as:

$$f(t) = \frac{L_i}{1 + exp\,(-\alpha(t - t_0))} \tag{1}$$

where $f(t)$ is the NEV share of light-duty vehicle stock in year t, L_i is the region-specific saturation ceiling, α is the diffusion rate, and t_0 is the inflection year. The model suits technology-substitution processes with early adoption, mass-market expansion, and saturation. The penetration series uses combined BEV + PHEV stock; Phase 2 then applies to BEV only.

Freely estimating L_i from early-stage data is unstable, since curve curvature is not yet identified. A structured ceiling is imposed instead:

$$L_i = L_0 \times \varphi_{income} \times \varphi_{public\ transit} \times \varphi_{cold} \times \varphi_{truck} \tag{2}$$

with ϕ_{truck} applied only in the U.S. Only α and t_0 are then fit by nonlinear least squares to each region's historical series. The base ceiling L_0 is 0.85 for China, 0.75 for the U.S. The lower U.S. base reflects higher annual vehicle kilometers travaled (VKT), sparser non-metro charging, and pickup/large-SUV dominance, a segment with constrained BEV availability and range tolerance.

Table 1. Saturation ceiling adjusters

Adjuster	Rationale	China brackets	US brackets
ϕ_{income}	Higher income widens addressable market	DPI[1] > ¥70K: 1.12 ¥70K > DPI > ¥50K: 1.05 DPI < ¥50K: 1.00	DPI > $75K: 1.12 $75K > DPI > $60K: 1.05 DPI < $60K: 1.00
$\phi_{public\ transit}$	Strong public transit dampens private demand	PTI[2] top 5: 0.85 PTI 6-25: 0.92 PTI rest: 1.00	PTU[3] top 5: 0.85 PTU 6-25: 0.95 PTU 26-50: 1.00
ϕ_{cold}	Cold reduces BEV range/utility	< 5°C:0.90 5-15°C: 0.95 > 15°C: 1.00	< 40°F: 0.90 40-55°F: 0.95 > 55°F: 1.00
ϕ_{truck}	Truck-heavy fleets resist electrification	n/a	T/A ratio[4] > 2.0 0.90 2.0 > T/A ratio > 1.5: 0.95 T/A ratio < 1.5: 1.00

Choice of model: the Fisher-Pry is selected over two common alternatives. The Bass diffusion model decomposes demand into innovation and imitation effects and is well-suited to predicting first-purchase take-off of a new product; it is less suited to modeling long-run fractional displacement of an incumbent technology, which is the object of interest here. Reliable Bass estimation also requires data spanning the inflection point of the adoption curve, which most provinces and states in this study have not yet reached as of 2024, leaving the imitation parameter poorly identified. Simple exponential extrapolation is rejected because it imposes no saturation ceiling and produces implausible long-run penetration rates. Fisher-Pry models the fractional replacement of one technology by another directly, requires only two free parameters once the ceiling is externally specified, and accommodates the structural prior on L_i described above, which is essential given the short, pre-inflection nature of the available series.

3.4 Phase 2: Per-Kilometer Emission Accounting

The net per-kilometer differential for region *i*, pollutant *p*, is:

[1] DPI: Disposable Personal Income, in an annual basis.
[2] PTI: Public Transit Index, calculated based on the sum of bus and rail traveller divided by total population of the specific province.
[3] PTU: Public Transportation Usage, measured by average miles per resident traveled via public transit per year.
[4] T/A ratio: Truck-to-Auto Ratio

$$\Delta E_{i,p} = E_{i,p}^{ICEV} - E_{i,p}^{BEV} \tag{3}$$

a positive value indicates BEV substitution reduces emissions. ICEV emissions = temperature-corrected tailpipe + tire wear; BEV emissions = power-plant emissions from charging + tire wear. The five steps below construct each term.

Step 1: Fleet emission-standard composition

The ICEV fleet is decomposed by emission-standard vintage, which sets the baseline tailpipe factor before temperature correction.

Table 2. Fleet composition method and result

	China	United States
Method	Iterative back-calculation from annual registrations using survival function	Direct model-year distribution from in-use fleet data
Survival function	4-segment, Hao et al. (2011), Zheng B. et al.(2019): 100% - 97% (fleet age 5), 80% (fleet age 10), 32% (fleet age 15), 3% (fleet age 16+)	n/a (observed distribution)
Source	CAIA stock-flow data	Hedges & Company (2024), BTS-derived
Standard mapping	GB18352.3 (2008), GB18352.4 (2011), GB18352.5 (2017), GB18352.6-2016a (2020), GB18352.6-2016b (2023)	Tier 0 (<1994), Tier 1 (1994–2003), Tier 2 (2004–2016), Tier 3 (2017+)
Result (fleet share)	GB18352.3: 4% GB18352.4: 31% GB18352.5: 25% GB18352.6-2016a/b: 39%	Tier 0: 3.6% Tier 1: 16.6% Tier 2: 52.2% Tier 3: 27.6%

The U.S. national distribution is applied uniformly to all states (no state-level fleet age data exists). The 2015-2019 U.S. cohort, spanning the Tier 2/3 transition at MY2017, is split proportionally across its five model years.

Step 2: Baseline tailpipe emission factors

Table 3. Tailpipe EF source and derivation

	China	United States
Standards source	MEE (2014) Guidelines, Table 5 (GB18352.1-5)	EPA rulemakings: pre-1990 (T0), 1990 CAA (T1), 1999 Rule (T2), 2014 Rule (T3)
Newest-standard derivation	GB18352.6-2016a/b extrapolated from GB18352.5, drawn from Vehicle Emission Control Center (2021) analysis presented to the Joint Research Centre	Class-weighted: LDV 45% + LDT1 35% + LDT2 20%
Adjustment	Embedded in MEE factors	Deterioration coefficients: T0 =

		1.00, T1 = 0.85, T2 = 0.55, T3 = 0.40 (EPA MOVES4, ICCT)
Unit	g/km	g/mile × 0.6214 = g/km

Resulting fleet-weighted tailpipe NOx is 0.026 g/km for China, 0.120 g/km for the U.S. The 5 times gap is driven by the older U.S. fleet (Tier 0 + 1: 20%, vs. China's GB18352.1-3: 4%) and historically looser early U.S. standards.

Step 3: Temperature correction

Tailpipe factors are scaled by climate, using MEE (2014) Table 8 coefficients applied to the share of the year in each temperature band:

Table 4. Temperature multipliers and band construction

Pollutant	Cold (<10°C / <50°F)	Mid (10–25°C / 50–77°F)	Hot (>25°C / >77°F)
CO	×1.36	×1.00	×1.23
HC	×1.47	×1.00	×1.08
NOx	×1.15	×1.00	×1.31
PM	×1.00	×1.00	×1.00

China: monthly distribution estimated from provincial annual means + regional seasonal amplitude (sinusoidal approximation).
US: NOAA monthly state temperatures used directly, each month classified into a band.

The same coefficient set is used in both countries for comparability. PM receives no correction (per MEE).

Step 4: Grid emission factors

BEV charging emissions use region-specific grid factors for CO_2, NOx, SO_2, PM.

Table 5. Grid EF source and derivation

	China	United States
CO_2	MEE (2024) provincial factors (2022 base)	eGRID 2023 ST23
NOx, SO_2	Derived: generation mix × ultra-low-emission limits (GB 13223)	eGRID 2023 ST23
PM	Derived alongside NOx/SO_2	Estimated (eGRID omits state PM), Eq. (4)
Unit	g/kWh	lb/MWh × 0.4536 = g/kWh

For China, coal generation is assumed at post-retrofit ultra-low limits (NOx 50, SO_2 35, PM 10 mg/m³), consistent with >94% of capacity retrofitted by 2022 (CEC, 2024). For the U.S., PM is estimated from fuel mix:

$$PM_s = \sum_f (Share_{s,f} \times EF_f^{PM}) \tag{4}$$

with PM2.5 factors (g/kWh): coal 0.090, oil 0.050, gas 0.007, biomass/other-fossil 0.100; PM10 scaled proportionally. This carries ±20% uncertainty but does not alter any state's tier under sensitivity testing.

National grid averages: CO_2 537 (China) vs. 347 g/kWh (US); the U.S. CO_2 spread across states is wider (19 g/kWh in VT to 887 in WV).

Step 5: Vehicle parameters and ΔE

Representative parameters are sales-weighted averages of 2025 top 50 selling models of ICEV and BEV in China and the United States (curb weights from manufacturer data; energy use from fueleconomy.gov / MIIT certification).

Table 6. Representative vehicle parameters

Parameter	China BEV	China ICEV	US BEV	US ICEV
Curb mass (kg)	1,581	1,603	2,106	1,833
Energy Efficiency	11.62 kWh/100km	6.45 L/100km	19.80 kWh/100km	9.04 L/100km
BEV - ICEV mass	-22 kg		+273 kg	

The mass relationship reverses between markets: parity in China (large micro-BEV share) versus a 15% heavier BEV fleet in the U.S. (large-SUV-dominated BEV mix). This directly affects the tire-wear comparison.

Tire wear uses the linear mass model (Giechaskiel et al., 2024):

$$TWP = 68 mg/km/tonne \times m \tag{5}$$

with PM2.5 = 1.0% and PM10 = 2.5% of TWP. ICEV CO_2 = fuel consumption × 2,310 gCO_2/L. BEV CO_2 = energy use × grid CO_2 factor.

3.5 Phase 3: Multi-Criteria Scoring and Tier Classification

Each region receives a composite score from three normalized dimensions (min-max to [0,100] within each country):

$$S_{env} = 0.5Norm(\Delta E_{CO2}) + 0.3Norm(\Delta E_{NOx}) + 0.2Norm(\Delta E_{PM2.5}) \tag{6}$$

$$S_{mkt} = Norm[(f_{2030} - f_{2024}) \times LDV_{2024}] \tag{7}$$

$$S_{base} = Norm(PM2.5_{2024}) \tag{8}$$

$$S_i = 0.4S_{env} + 0.35S_{mkt} + 0.25S_{base} \tag{9}$$

S_{env} (40%): the core environmental return.
S_{mkt} (35%): absolute BEV stock increment, not percentage, so high-stock regions weight appropriately.
S_{base} (25%): higher baseline pollution lead to higher marginal health benefit. China PM2.5 data from MEE air quality reports; US data from EPA County Factbook 2024, county-population-weighted to state.

Tier classification uses thresholds $S_{env} = 50$, $S_{mkt} = 30$ as quadrant boundaries, with S_{base} differentiating within quadrants.

Table 7. Tier definitions

Tier	Condition	Policy direction
1	$S_{env} > 50$, $S_{mkt} > 30$	Priority deployment: benefit + scale
2	$S_{env} > 50$, $S_{mkt} \leq 30$, high baseline	Targeted fleet electrification
2b	$S_{env} > 50$, $S_{mkt} \leq 30$, low baseline	High benefit, lower health urgency
3	$S_{env} \leq 50$, high baseline	Adverse local pollutant, but AQ already poor
4	$S_{env} \leq 50$, low baseline	Adverse outcome, clean air: defer / grid-first
5	No standout dimension	Low priority

4. Result

4.1 Phase 1: Adoption Trajectories to 2030

Fitting Equation (1) to each region's 2017-2024 series yields strong fits across both countries (mean $R^2 = 0.97$ China, 0.98 U.S.; all regions $R^2 > 0.90$). Table 8 summarizes the leading and lagging regions.

Table 8. 2030 NEV penetration forecast, selected regions

	Top 5 (f_{2030})	Bottom 5 (f_{2030})
China	Jiangsu, Hainan, Zhejiang, Sichuan, Guangdong (all > 50%)	Xinjiang, Tibet, Inner Mongolia, Qinghai, Gansu (all < 35%)
US	Delaware 38%, DC 33%, Nevada 29%, New Jersey 27%, California 26%	North Dakota, South Dakota, Wyoming, Mississippi, Louisiana (all < 5%)

Two cross-national patterns stand out. First, U.S. adoption trails China by roughly five to six years: the highest-penetration U.S. state in 2030 (Delaware, 38%) sits below China's national average trajectory. Second, the within-country spread is far wider in the U.S. (a roughly 25-fold gap between top and bottom states) than in China (a roughly 2-fold gap across provinces), reflecting the U.S.'s sharper divergence between coastal-urban and rural-interior markets.

4.2 Phase 2: Per-Kilometer Environmental Impact

ΔCO_2: BEV substitution reduces CO_2 in every region of both countries, but the magnitude varies sharply with grid composition. China ranges from +56 g/km (Inner Mongolia) to +137 g/km (Yunnan); the U.S. ranges from +33 g/km (West Virginia) to +205 g/km (Vermont). The U.S. mean (142 g/km) exceeds China's (88 g/km) despite the dirtier average U.S. grid, because the heavier, higher- consumption U.S. ICEV baseline emits more CO_2 per km to begin with.

ΔNOx: the central cross-national divergence. The two countries diverge fundamentally on local pollutants. In China, BEV substitution worsens NOx in 16 of 31 provinces, as power-plant NOx in coal-heavy grids exceeds the tailpipe NOx of a relatively modern ICEV fleet. In the U.S., BEV substitution reduces NOx in all 51 states. The explanation is the fleet-age gap documented in Step 2: the U.S. fleet's tailpipe NOx baseline (0.120 g/km) is roughly five times China's (0.026 g/km), so even the dirtiest U.S. grid does not generate enough power-plant NOx to offset the displaced tailpipe emissions.

ΔPM2.5 and tire wear: the mass reversal from Table 6 surfaces here. In China, tire wear PM roughly canceled due to mean BEV and ICEV curb mass, and ΔPM2.5 is positive in all provinces. In the U.S., the 15%-heavier BEV fleet generates more tire wear, and combined with coal-grid power-plant PM, ΔPM2.5 turns negative in five coal-dependent states (West Virginia, Kentucky, Wyoming, Missouri, Indiana), where BEV substitution increases fine particulate emissions.

Table 9. Phase 2 cross-national summary

Metric	China	United States
ΔCO_2 range (g/km)	+56 to +137	+33 to +205
ΔCO_2 mean	88	142
Provinces/states where ΔNOx < 0	16 of 31	0 of 51
Provinces/states where ΔPM2.5 < 0	0 of 31	5 of 51
BEV vs ICEV mass (kg,%)	-22, 98.6%	+273, 114.9%

4.3 Phase 3: Composite Scores and Tier Classification

Applying Equations (6)–(9) produces the tier distribution in Table 10.

Table 10. Tier distribution

Tier	Policy direction	China (31)	US (51)
1	Priority deployment: benefit + scale	3 (Sichuan, Guangdong, Guangxi)	3 (California, Texas, Florida)
2	Targeted fleet electrification	4	17
2b	High benefit, lower health urgency	5	22
3	Adverse local pollutant, but AQ already poor	4	0
4	Adverse outcome, clean air: defer /	1	0

	grid-first		
5	Low priority	14	9

The defining structural difference appears in Tiers 3 and 4. China places five provinces there: regions where BEV substitution carries a local-pollutant penalty, requiring coordination with grid decarbonization before aggressive deployment. The U.S. has none, a direct consequence of the all-positive ΔNOx result in 4.2. The older U.S. fleet means BEV substitution never produces a net local-pollutant penalty severe enough to drop a state below the environmental threshold, so the U.S. policy problem is one of prioritization (which states first) rather than conditionality (where substitution may backfire).

California's dominance of U.S. Tier 1 (composite score 95.8, far above Texas and Florida) reflects its simultaneous strength on all three dimensions: high environmental benefit, the largest BEV market increment, and the most polluted baseline among major states. In China, the leading provinces combine large markets with at least moderate environmental benefit, while the highest per-km benefit provinces (Yunnan, Sichuan, Qinghai) rank lower on the composite where their market scale is small.

5. Policy Analysis and Discussion

5.1 The Inefficiency of Uniform Incentives

The results in Section 4 establish that the environmental return to BEV substitution varies by a factor of 2 to 6 across regions within each country, driven by grid composition and fleet structure. This variation has a direct fiscal implication: when the per-kilometer environmental benefit differs across regions, an incentive of equal magnitude purchases unequal environmental improvement depending on where it is spent. A subsidy that displaces an ICEV in Vermont removes roughly 6 times the CO_2 per kilometer of the same subsidy spent in West Virginia, and in the five coal-dependent U.S. states where ΔPM2.5 is negative, the same fiscal outlay can worsen local particulate pollution while reducing carbon. A uniform incentive is efficient as environmental policy only if the environmental return is uniform, but the evidence shows it is actually not. The two countries depart from this efficiency benchmark in different ways.

China's incentive structure is converging toward nationally uniform parameters: the 2026 trade-in framework replaced the prior provincial patchwork with a single price-proportional formula applied identically across all provinces (Section 2.2.3). This delivers administrative simplicity and market unification, but it embeds the misallocation problem directly into national policy, since the same percentage subsidy flows to coal-grid Inner Mongolia and hydropower-rich Yunnan alike. China's challenge is therefore one of uniform-but-misallocated incentives.

The United States presents a different problem. Even under the IRA, the federal credit was only one layer of a system in which the larger and more durable incentives operated at the state level, through ZEV mandates and state purchase rebates. With the federal credit terminated in 2025, the U.S. incentive landscape is now predominantly state-driven and, critically, less resolute: federal policy has reversed sharply with changes in administration, undermining the continuity and predictability that long-horizon vehicle-purchase and infrastructure decisions require. The U.S. challenge is therefore one of volatility and fragmentation rather than China's uniform misallocation.

5.2 A Differentiated Policy Framework

The tier classification from Phase 3 converts directly into a differentiated instrument assignment. Table 11 maps each tier to a policy package.

Table 11. Tier-to-instrument matrix

Tier	Subsidy intensity	Charging infrastructure	Public-fleet electrification	Grid coordination
1	Full	Aggressive build-out	Lead by example	Standard
2	Moderate, fleet-focused	Depot/transit corridors	Primary lever	Standard
2b	Low, signaling only	Market-led	Optional	Standard
3	Conditional	Build-out	Yes	Precondition required
4	Defer	Defer	Defer	Grid-first
5	Minimal	Market-led	Optional	Standard

The central design principle is that subsidy intensity should track its environmental return, and that in Tier 3 and Tier 4 regions, vehicle-side incentives should be sequenced after or alongside grid decarbonization rather than deployed independently, since BEV substitution there carries a net local-pollutant penalty until the grid cleans.

A practical objection concerns financing: differentiated subsidies that vary by region are administratively complex and politically fraught if funded from general fiscal revenue, particularly after the U.S. federal credit's repeal removed the primary funding channel. Here China's Dual Credit mechanism offers a transferable model. Rather than financing incentives from the public budget, the Dual Credit system requires high-emission manufacturers to purchase NEV credits from low-emission producers, creating an intra-industry subsidy that operates independently of fiscal appropriations. A U.S. analogue, a tradable-credit obligation administered at the federal or multi-state level, could sustain BEV incentives through the current period of federal fiscal withdrawal without recourse to congressional appropriations, while a regionally weighted credit value could in principle direct stronger incentives toward higher-return regions. This would address both the U.S. continuity problem and the financing question simultaneously.

5.3 Country-Specific Implications

For China, three implications follow:

- First, the coal-dependent northern provinces (Inner Mongolia, Shanxi, and the broader Tier 3/4 group) should treat grid decarbonization as a precondition for aggressive BEV promotion; subsidizing BEV uptake on a coal-heavy grid delivers limited carbon benefit and a local-pollutant penalty.
- Second, the high-market, high-pollution-baseline provinces such as Hebei and Henan are underweighted by a uniform policy: their large vehicle stocks and bad air quality make them high-value targets despite moderate per-kilometer benefit.
- Third, the clean-grid western provinces (Yunnan, Qinghai, Sichuan) offer the highest per-kilometer benefit but limited market scale, policies for them should suit for demonstration and public-fleet roles rather than mass-market subsidy.

For the United States, also three implications follow:

- First, with federal incentives withdrawn, the burden of any differentiated policy now falls on states and on the kind of intra-industry mechanism described above; states with strong environmental returns but no state-level incentive program face the steepest cliff.
- Second, the absence of any Tier 3/4 state is itself a policy-relevant finding: because the aged U.S. fleet means BEV substitution reduces NOx everywhere, the U.S. faces no "wrong place to electrify" problem on local pollutants, and policy can pursue deployment more uniformly than in China, subject to prioritization.
- Third, the five coal-state exception on ΔPM2.5 (West Virginia, Kentucky, Wyoming, Missouri, Indiana) warrants the same grid-coordination sequencing recommended for China's Tier 3, even though these states do not fall below the composite environmental threshold.

5.4 Cross-National Lessons

The two systems have complementary strengths. China's Dual Credit mechanism demonstrates how to finance electrification incentives without relying on the fiscal appropriations that have proven politically fragile in the United States, and its recent move to nationally uniform parameters shows the administrative appeal, but also the misallocation cost, of simplicity. The U.S. experience, in turn, illustrates the value of durable sub-national policy instruments such as the California ZEV mandate, which have outlasted multiple reversals of federal policy. The common lesson is that an environmentally efficient incentive design must be both spatially differentiated, to match the heterogeneity of environmental returns documented here, and institutionally durable, to provide the continuity that vehicle and infrastructure investment decisions require. Neither country's current framework satisfies both conditions simultaneously.

6. Limitations and Future Work

Several limitations qualify the findings and indicate directions for further research:

- Use-phase boundary: the analysis excludes manufacturing and end-of-life emissions (Section 3.1). BEV production is more carbon-intensive than ICEV production, primarily due to battery manufacturing, so the use-phase comparison reported here is favorable to BEVs relative to a full lifecycle accounting. The use-phase focus is appropriate for the geographic policy question addressed, but the results should not be read as full lifecycle verdicts.
- Static parameters: grid emission factors, fleet composition, and vehicle parameters are held at their most recent observed values. Both grids are decarbonizing, particularly China's, so the BEV advantage will widen over the forecast horizon in ways the static calculation does not capture. The 2030 adoption forecast is likewise sensitive to policy reversals that a diffusion model cannot anticipate, a limitation made concrete by the 2025 U.S. federal credit repeal.
- Estimation assumptions: three approximations carry quantifiable uncertainty. U.S. state-level PM emission factors are estimated from fuel mix rather than measured directly; the U.S. fleet age distribution is applied uniformly across states for lack of state-level data; and the temperature correction for BEV energy consumption was not applied, which understates winter efficiency losses in cold regions and modestly overstates BEV benefit in the coldest provinces and states. Sensitivity testing indicates none of these alter the tier classification of any region, but they affect point estimates.
- Cross-province and cross-state electricity flows: grid factors are assigned on a generation basis within each region. Net electricity importers, such as Beijing, Shanghai, and several northeastern U.S. states, are therefore assigned the emission intensity of their own generation rather than of the power they actually consume, which may misstate their true BEV charging emissions.

Three extensions would strengthen the analysis in future work: incorporating consumer heterogeneity and price elasticity to estimate the actual behavioral response to differentiated subsidies; monetizing the health impacts of the local-pollutant changes to place CO_2 and air-quality effects on a common welfare scale; and modeling the joint trajectory of grid decarbonization and BEV adoption to capture the dynamic interaction that the static framework omits.

7. Conclusion

This study evaluated the per-kilometer environmental impact of substituting battery electric vehicles for internal combustion engine vehicles across 31 Chinese provinces and 51 U.S. states, using a harmonized three-phase framework spanning adoption forecasting, multi-pollutant emission accounting, and multi-criteria policy scoring.

The central finding is that the environmental return to BEV substitution is highly heterogeneous across subnational regions, and that the structure of this heterogeneity differs fundamentally between the two countries. BEV substitution reduces CO_2 everywhere in both countries, but by a factor that varies 2 to 6 fold with local grid composition. On local pollutants the two countries diverge sharply: BEV substitution worsens NOx in 16 of 31 Chinese provinces, yet reduces NOx in all 51 U.S. states, a divergence traceable to the much older U.S. vehicle fleet and its correspondingly higher tailpipe baseline. The sales-weighted vehicle data also overturn a common assumption, showing near mass-parity between BEVs and ICEVs in China but a 15% heavier BEV fleet in the United States, with the latter producing negative ΔPM2.5 in five coal-dependent states.

These findings carry a consistent policy implication. The uniform-subsidy designs that have prevailed in both countries are inefficient as environmental policy, because they spend equal resources for unequal environmental return. The differentiated framework proposed here aligns subsidy intensity, infrastructure investment, and grid-coordination requirements with each region's measured environmental benefit, market scale, and air-quality baseline. The two countries face different versions of the problem: China's incentives are converging toward nationally uniform parameters that embed spatial misallocation, while the United States faces federal policy volatility that undermines the continuity sub-national differentiation requires. China's Dual Credit mechanism suggests a financing model capable of sustaining differentiated incentives without fiscal appropriations, and the California ZEV mandate illustrates the durability that effective sub-national instruments can achieve. An environmentally efficient incentive regime must be both spatially differentiated and institutionally durable. Neither country's current framework meets both conditions, and closing that gap is the central task this analysis identifies for EV policy in both countries.

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