

Drivers and implications of declining fossil fuel CO₂ in Chinese cities revealed by radiocarbon measurements

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Abstract. China's clean air policies have successfully mitigated fossil fuel CO₂ (CO_{2ff} or C_{ff}) emissions in bottom-up inventories since 2013. Yet, evidence from top-down measurements and their underlying drivers remains lacking. Here, we quantify C_{ff} concentrations and fuel-specific contributions using atmospheric $A(^{14}\text{CO}_2)$ and $\delta(^{13}\text{CO}_2)$ measurements across representative Chinese cities. We found distinct regional trends: megacities like Guangzhou show significant C_{ff} declines (34–46% decrease from 2010 to 2022) along with their source regions, while smaller cities have yet to demonstrate similar reductions. These improvements can be attributed to a 23 % coal consumption reduction, 17 % increased natural gas use (evidenced by stable isotope analysis), and improved combustion efficiency (indicated by 63 % falling R_{CO/CO_{2ff}} ratios). Notably, the three-decade observational record shows steeper declines in urban R_{CO/CO_{2ff}} ratios than inventory estimates, suggesting current emission inventories may underestimate combustion efficiency improvements and CO emission reductions relative to C_{ff} mitigations. These findings indicate nationwide progress toward C_{ff} emission peaks, with megacities leading the transition. They also underscore how coal-to-gas transitions and technological upgrades simultaneously advance air quality and climate goals. Importantly, our results highlight the critical need to integrate top-down observational frameworks (e.g. radiocarbon measurements) with traditional inventories to better capture rapid, policy-driven emission changes and inform future co-benefit optimization strategies.

35 1 Introduction

As the world's largest energy consumer, China's heavy reliance on fossil fuels has resulted in severe air pollution and substantial fossil fuel CO₂ (CO_{2ff} or C_{ff}) emissions, accounting for 31 % of global fossil CO₂ emissions in 2022 (Friedlingstein et al., 2023a). These emissions pose critical threats to public health and ecological stability. In response, China has enacted progressive policies including the 2013 Clean Air Action Plan (Zheng et al., 2018; Zhang et al., 2019), 40 2018 Blue Sky Defense Battle, and 2022 Pollution-Carbon Synergy Plan, achieving co-benefits in air quality improvement and C_{ff} mitigation as quantified through bottom-up inventories like Multi-resolution Emission Inventory for China (MEIC) (Shi et al., 2022). However, the effectiveness of these policies in reducing atmospheric C_{ff} concentrations, and the underlying drivers of these reductions, remains unverified and unexplored through top-down observational approaches, creating a critical knowledge gap in climate policy assessment.

45 Bottom-up inventories and top-down measurements are approaches commonly used to determine atmospheric C_{ff} emissions, but each has inherent limitations that can affect accuracy and reliability. Although bottom-up inventories are available at increasingly higher spatiotemporal resolution (Han et al., 2020), they are time-consuming to compile and update promptly, often lack quantitative estimation of uncertainty (Lo Vullo and Monforti, 2019), and frequently debated in attributing emissions to specific sources (Gurney et al., 2021). In contrast, top-down studies encompass all existing sources within a 50 geographic region but struggle to achieve accurate partitioning of the fossil fuel and biospheric CO₂ contributions. This methodological impasse can be resolved by ¹⁴C analysis, which exploits the unique ¹⁴C-depletion signature of C_{ff} compared to contemporary biogenic sources (Levin et al., 2003; Turnbull et al., 2006), enabling unambiguous fossil fuel emission quantification.

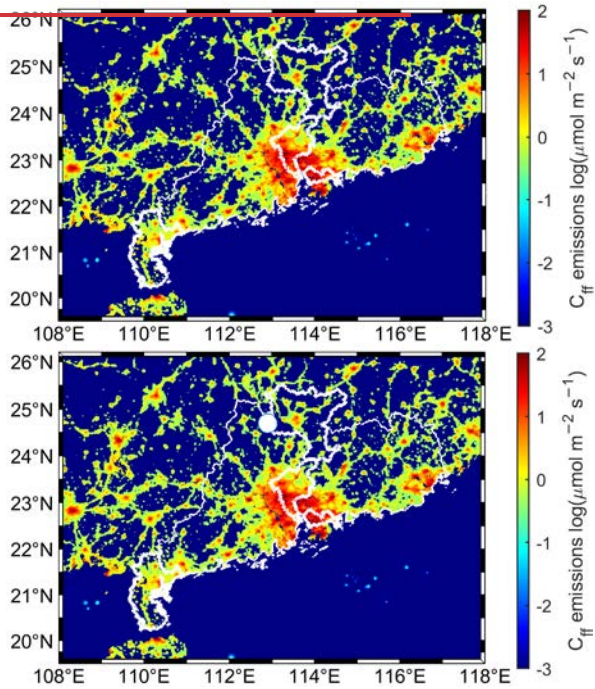
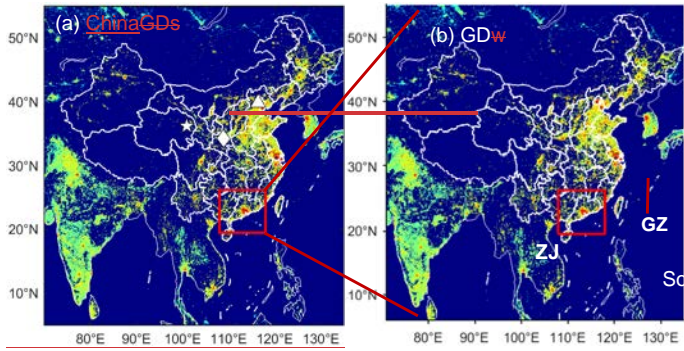
Urban areas, occupying merely 3 % of global land yet responsible for 75 % of global C_{ff} emissions (reaching 80 % in China) 55 (Dhakal, 2009; Duren and Miller, 2012), represent strategic priorities for emission mitigation. Recent advances in analytical tools can help identify key drivers of urban C_{ff} reductions. $\delta(^{13}\text{CO}_2)$ signatures successfully distinguished coal, oil, and natural gas contributions in cities like Beijing and Xi'an (Wang et al., 2022b), while $\Delta\text{CO}/\text{C}_{\text{ff}}$ (ΔCO denotes the difference between observed and background values; $\Delta\text{CO} = \text{CO}_{\text{obs}} - \text{CO}_{\text{bg}}$) ratios tracked combustion efficiency variations across national (China, South Korea) and urban (Paris, Heidelberg) scales (Turnbull et al., 2011; Lee et al., 2020; Lopez et al., 2013; 60 Rosendahl, 2022). To address the research gaps mentioned above, we performed spatiotemporal mapping of 2022 C_{ff} concentrations across representative Chinese cities using dual-carbon isotope constraints ($\Delta(^{14}\text{CO}_2) + \delta(^{13}\text{CO}_2)$) for fuel-specific source attribution. By integrating multi-source inventories with extended $\Delta\text{CO}/\text{C}_{\text{ff}}$ observations through 2022, we developed a robust framework for top-down verification of policy-driven emission reductions. Our methodology not only quantifies C_{ff} concentration decreases but also identifies the key mechanisms behind these reductions, offering critical 65 insights for refining climate mitigation strategies and supporting sustainable urban development.

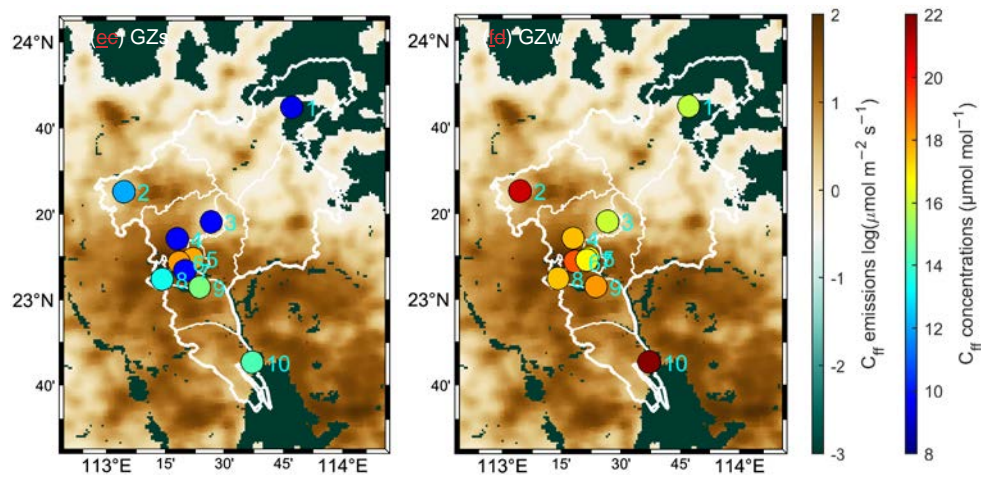
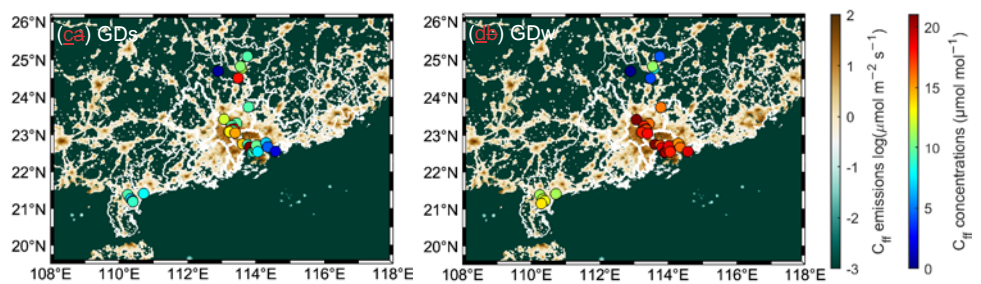
2 Data and methods

2.1 Study area and sample collection

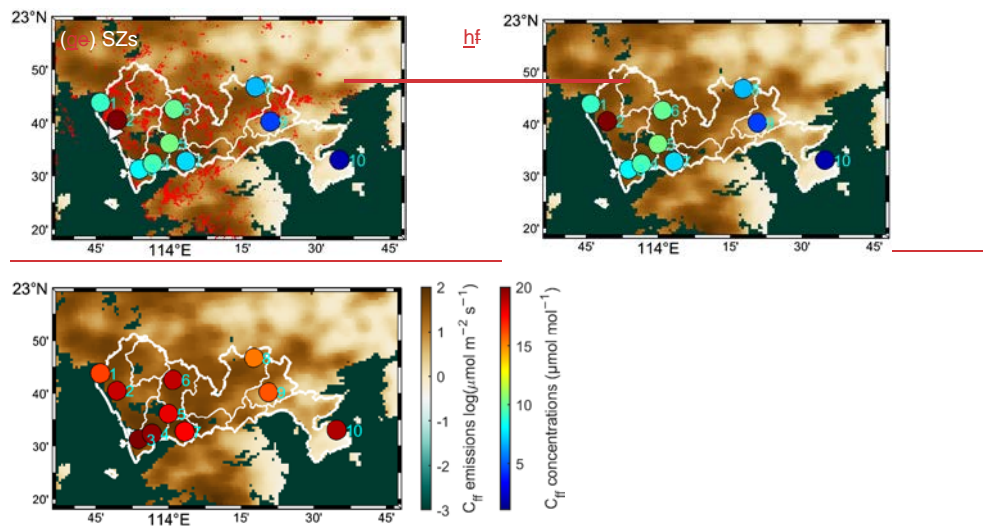
We selected representative Chinese cities of varied population sizes for this study: Guangzhou, Shenzhen, and Beijing for megacities (urban permanent resident populations >10 million), Xi'an for supercities (5–10 million), Zhanjiang for large cities (1–5 million), and Shaoguan for medium and small cities (<1 million), which is retrieved from the Tabulation on 2020 China Population Census by County (Council, 2022). Since we could obtain results in Beijing and Xi'an from previous studies, we conducted field sampling in the four cities in Guangdong Province, China (Fig. 1). Guangdong Province is located south of the Nanling Mountains and on the coast of the South China Sea, lying within subtropical and tropical low-latitude regions. The area experiences a prevailing southeast monsoon from the ocean during summer and a northeast monsoon from the continent during winter. The four cities in Guangdong Province differ in terms of area, population, gross domestic product (GDP), C_{ff} inventory emissions, population density, topographic elevation, and land use/land cover. Guangzhou and Shenzhen represent two of China's seven megacities — approximately 45 exist globally — within the Pearl River Delta (PRD), the world's largest urban agglomeration. Guangzhou, the capital of Guangdong Province, has a population of 18.7 million, GDP of 2 884 billion Yuan, and built-up area covering 35.2 %. Shenzhen, a high-tech hub transformed by post-1978 reforms, hosts 17.7 million people with GDP reaching 3 239 billion Yuan and 53.8 % built-up coverage. In contrast, Zhanjiang (large city) and Shaoguan (medium and small city) have smaller populations — 7.0 million and 2.9 million respectively — and lower GDPs of 371.3 billion Yuan and 156.4 billion Yuan. Zhanjiang features extensive cultivated land (47.8 %) and coastal ports, while Shaoguan is distinguished by 78.4 % vegetation coverage.

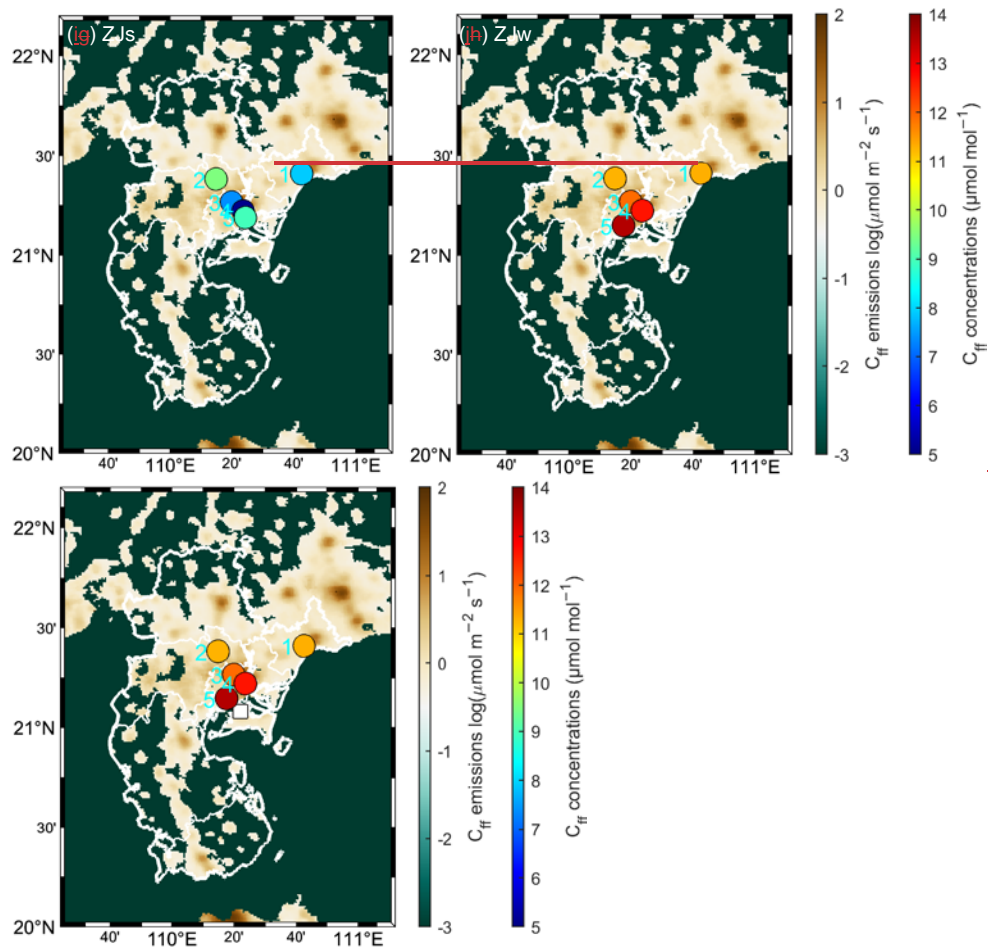
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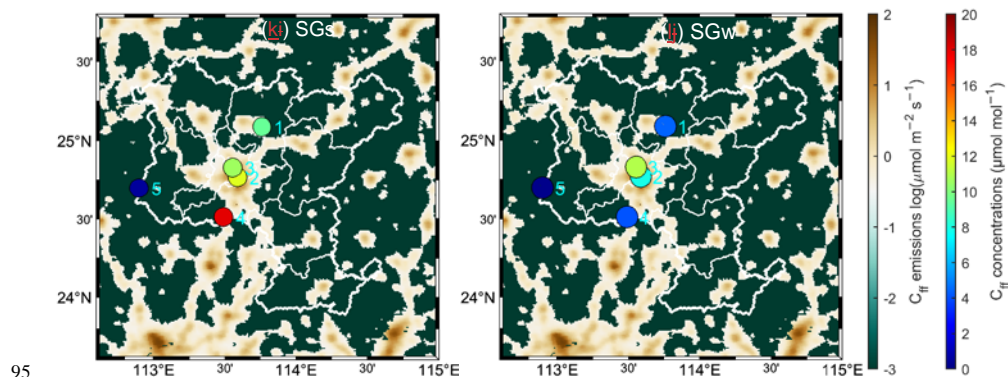


Figure 1: Locations of sampling sites and spatial distribution of C_{ff} concentrations during summer (s) and winter (w) in (ca and db, GD) Guangdong Province and the cities of (ee and fd, GZ) Guangzhou, (ge and hf, SZ) Shenzhen, (ig and jb, ZJ) Zhanjiang, and (ki and lj, SG) Shaoguan. White-filled symbols denote Beijing (\blacktriangle), Xi'an (\blacklozenge), and Waliguan (\blackstar) in (a); Nanling (\bullet) in (b); Shenzhen Airport (\blacktriangleright) in (g); and Zhanjiang Port (\blacksquare) in (i).— Shenzhen's industrial land use (<https://download.geofabrik.de/asia/china.html>, last access: 11 Nov. 2025) is shown as red in (g), with the spatial distribution of all industrial enterprises and CO₂-emitting facilities documented in Li et al. (2025a). The colored circles in (c-l) represent the observations, while the shading indicates the C_{ff} inventory emissions from the Open-source Data Inventory for Anthropogenic CO₂ (ODIAC) (Oda and Maksyutov, 2024) in August and December with 1×1 km grid spacing. White lines indicate boundaries of cities in Guangdong Province (ca and db), and boundaries of districts in the four cities (ee–lj). In (ca) and (db), bold white lines indicate boundaries of nine cities of the Pearl River Delta. The left color bar represents the C_{ff} inventory emissions, while the right color bar represents the C_{ff} observations.

We collected 240 air samples from 30 sites during summer (28 July – 30 August 2022) and winter (12 December 2022 – 6 January 2023) campaigns, with weekly sampling in both periods. Because atmospheric transport variability can influence observed C_{ff} signals, we evaluated the meteorological representativeness of the sampling months using ERA5 diagnostics and trajectory analyses. Specifically, we assessed whether the August and December 2022 flask measurements were representative of typical summer and winter transport conditions. Standardized anomalies ($z = (x_{target} - \mu_{season}) / \sigma_{season}$) were calculated for five ERA5 meteorological variables: 10 m eastward wind (U10), 10 m northward wind (V10), 2 m air temperature (T2M), surface pressure (SP), and planetary boundary-layer height (PBLH). Each target month was compared against (i) the concurrent 2022 seasonal background (June–July–August, JJA; December–January–February, DJF) and (ii) the 2010–2021 seasonal climatology. The choice of 2010 as the starting year ensures consistency with the earlier dataset from 2010, which is directly compared in this study. A month was considered “typical” when $|z| < 1$ and its dominant wind direction fell within the canonical summer (90–225°) or winter (0–45°) monsoon sectors.

The locations and details of these sampling sites are shown in Fig. 1 and summarized in Table A1. Ten sampling sites were located in Guangzhou (GZ1–GZ10), ranging from urban downtown to suburban areas, selected based on spatial gradients distributed of C_{ff} emissions derived from the Open-source Data Inventory for Anthropogenic CO₂ (ODIAC) (Oda

and Maksyutov, 2011; Oda and Maksyutov, 2024). Another 10 sampling sites were distributed uniformly throughout Shenzhen (SZ1–SZ10). In Zhanjiang (ZJ1–ZJ5) and Shaoguan (SG1–SG5), five sampling sites were selected in each city, primarily in urban areas, and distributed according to the first and second most dominant wind directions. These sites are located on the tower or on the roof of the building with 10–12 m extendable masts and are chosen to be free from any modifying effects of surrounding skyscrapers. Most of our sampling sites are generally no more than 300 m from the nearest air quality monitoring station. The sampling height is usually kept above 30 m above the ground level to avoid the influence of local sources. We assume that the measurements at the sampling sites in Guangzhou and Shenzhen are statistically sufficient to assess the whole cities, while those in Zhanjiang and Shaoguan are sufficient to assess the urban areas. Air sampling occurred between 13:00 and 17:00 local time, coinciding with the deepest planetary boundary layer and well-mixed atmospheric conditions. Post-filtration samples were transferred into pre-evacuated/flushed 3 L borosilicate flasks using 12 V micro-diaphragm pumps. These delivered a flow rate of 6 L min⁻¹ at 25 °C and 101.3 kPa, with pressurization to 172.4–206.8 kPa. The duration of the sampling was approximately 15–20 min in total.

2.2 Measurement of atmospheric CO₂ and δ(¹³C)

Whole-air samples were dried using magnesium perchlorate at a constant flow rate of 25 mL min⁻¹, controlled by a mass flow controller. The CO₂ concentrations and δ(¹³C) values were then measured using a Picarro G2201-i high-precision carbon isotope analyzer (Picarro, Inc., Santa Clara, CA, USA) with cavity ring-down spectroscopy. Each sample was measured for 10 min, and only data from the final 5 min were used to calculate the average CO₂ concentration and δ(¹³C) value. Calibration for the CO₂ concentrations and the δ(¹³C) values was performed using the method described by Wen et al. (2013) with three standards: (a) (409.47 ± 0.02) ppm (μmol mol⁻¹; similar hereafter), (-8.717 ± 0.013) ‰; (b) (447.78 ± 0.01) μmol mol⁻¹, (-9.759 ± 0.006) ‰; and (c) (503.65 ± 0.01) μmol mol⁻¹, (-11.456 ± 0.004) ‰, obtained from the Chinese Academy of Meteorological Sciences. The CO₂ concentrations of the standards are traceable to the X2019 standard scale maintained by the Central Calibration Laboratory of the World Meteorological Organization, and the δ(¹³C) values are traceable to the stable isotope laboratory of the Institute of Arctic and Alpine Research based on the NBS-19 and NBS-20 standards. The δ(¹³C) values were reported relative to the international Vienna Pee Dee Belemnite standard (Coplen, 1996). The precision was better than 0.2 μmol mol⁻¹ for CO₂ concentrations and 0.1 ‰ for δ(¹³C) values.

2.3 Measurement of atmospheric Δ(¹⁴C)

The residual air samples were transferred into a vacuum system at a flow rate of 300 mL min⁻¹. It was then first passed through a cold trap consisting of dry ice and ethanol slurry to freeze out water, followed by passage through a liquid nitrogen cold trap (-196 °C) to freeze down the CO₂ (Xu et al., 2007). The extracted and purified CO₂ was converted into graphite using the hydrogen reduction method. The graphite was then pressed into an aluminum holder for ¹⁴C measurements using

an NEC 0.5MV 1.5SDH-2 accelerator mass spectrometer (AMS, National Electrostatics Corporation, USA) (Zhu et al., 2015). Each measurement wheel typically comprises 13 primary standards (oxalic acid II), 13 secondary standards (IAEA-C7), 13 solid process blanks (*p*-phthalic acid), 6 gas process blanks (¹⁴C-free CO₂ in synthetic air from a cylinder), and some authentic air samples. The results are presented as $\Delta(^{14}\text{C})$, which is the per mill (‰) deviation from the absolute radiocarbon reference standard, corrected by fractionation and decay (Stuiver and Polach, 1977). We analyzed 17 pairs of parallel air samples to evaluate the quality control and assurance of the entire sampling and laboratory analysis process, including sampling, extraction, graphitization, and AMS measurement. The AMS measurement uncertainty and the average deviation are $(2.1 \pm 0.3) \text{‰}$ and $(0.2 \pm 2.9) \text{‰}$, respectively (see Fig. A1). We thus specify a one-sigma measurement uncertainty of 2.9 ‰ for $\Delta(^{14}\text{C})$ based on these repeat measurements of air samples.

2.4 C_{ff} concentration estimation (incorporated biomass burning emissions)

Recently added atmospheric CO₂ (CO_{2obs} or C_{obs}) is thought to consist of background CO₂ (CO_{2bg} or C_{bg}) and excess CO₂ (CO_{2xs} or C_{xs}). The C_{xs} mainly includes C_{ff} and biogenic CO₂ (CO_{2bio} or C_{bio}). The corresponding $\Delta(^{14}\text{C})$ values are expressed as Δ_{obs} , Δ_{bg} , Δ_{ff} (−1000 ‰, zero ¹⁴C content), and Δ_{bio} , respectively. The mass balance equations for atmospheric CO₂ and $\Delta(^{14}\text{C})$ are expressed as follows (Levin et al., 2003):

$$C_{\text{obs}} = C_{\text{bg}} + C_{\text{xs}} = C_{\text{bg}} + C_{\text{ff}} + C_{\text{bio}} \quad (1)$$

$$C_{\text{obs}}\Delta_{\text{obs}} = C_{\text{bg}}\Delta_{\text{bg}} + C_{\text{ff}}\Delta_{\text{ff}} + C_{\text{bio}}\Delta_{\text{bio}} \quad (2)$$

$$C_{\text{ff}} = \frac{C_{\text{obs}}(\Delta_{\text{bg}} - \Delta_{\text{obs}})}{\Delta_{\text{bg}} + 1000\text{‰}} - \frac{C_{\text{bio}}(\Delta_{\text{bg}} - \Delta_{\text{bio}})}{\Delta_{\text{bg}} + 1000\text{‰}} = \frac{C_{\text{obs}}(\Delta_{\text{bg}} - \Delta_{\text{obs}})}{\Delta_{\text{bg}} + 1000\text{‰}} - \beta \quad (3)$$

The added C_{ff} component is determined using Eq. (3). The CO₂ and $\Delta(^{14}\text{C})$ from other sources, such as air-sea exchange (see Appendix C1)(Graven et al., 2018) and nuclear facilities (see Appendix C2), have been neglected owing to their relatively small amounts. The second term (β) represents a disequilibrium correction for the effect of CO₂ sources from biospheric exchange with distinct $\Delta(^{14}\text{C})$ signatures relative to atmospheric values, primarily attributed to heterotrophic respiration (Rh) and biomass burning (BB). We quantified β using integrated modeling frameworks (see Appendixes B and C3). The heterotrophic respiration correction (β_{RH}) was derived from FLEXPART simulations (Pisso et al., 2019) with CASA-GFED4s data (Randerson et al., 2017; Van Der Werf et al., 2017), yielding values of $(-0.06 \pm 0.03) \mu\text{mol mol}^{-1}$ in summer and $(-0.11 \pm 0.04) \mu\text{mol mol}^{-1}$ in winter. The biomass burning corrections (β_{BB}) was calculated under two assumptions: (1) $\Delta(^{14}\text{C})$ endmembers assume 100 % perennial biomass, and (2) C_{BB} emissions represent 100 % of C_{bio} in EDGAR2024 (covering open and closed combustion) (Edgar, 2024). β_{BB} showed maximum values of $(-0.09 \pm 0.08) \mu\text{mol mol}^{-1}$ during summer and $(-0.24 \pm 0.12) \mu\text{mol mol}^{-1}$ during winter. The combined correction ($\beta = \beta_{\text{RH}} + \beta_{\text{BB}}$) under the maximum-assumption simulation was displayed seasonal patterns: $(-0.16 \pm 0.09) \mu\text{mol mol}^{-1}$ in summer and $(-0.35 \pm 0.15) \mu\text{mol mol}^{-1}$ in winter, which contrast~~s~~ing with the seasonal pattern in Turnbull et al. (2009): $(-0.5 \pm 0.2) \mu\text{mol mol}^{-1}$ during summer and $(-0.2 \pm 0.1) \mu\text{mol mol}^{-1}$ during winter reported by Turnbull et al. (2009). Notably, This study is the first to integrate BB

emissions into C_{ff} estimation frameworks, revealing its dominant impact over than Rh on disequilibrium corrections under these, likely due to inventory and assumption differences. While the total corrections align broadly with literature values, the seasonal trends differed inversely from past studies. For the final C_{ff} estimations, we applied literature-based corrections from Turnbull et al. (2009) to maintain methodological consistency and comparability with previous work, and the simulated broad comparability ($< -0.5 \mu\text{mol mol}^{-1}$) in the magnitude of the corrections provides independent support for the applied values, and our simulations confirm that the main conclusions are robust across this range of potential corrections. disequilibrium terms further support our conclusions.

2.5 C_{ff} footprint by FLEXPART model

Surface flux sensitivity of C_{ff} were conducted using the FLEXible PARTicle (FLEXPART) dispersion model (version 10.4) (Pisso et al., 2019). The model produced source-receptor relationships, often referred to as “footprints” for atmospheric surface measurements, which represent the response of the observations at a measuring station to a source emission. The footprints are calculated driven by global meteorological fields from the National Centers for Environmental Prediction’s Climate Forecast System (CFSv2) Reanalysis model (Saha et al., 2011). They are computed by releasing 10 000 virtual particles from each receptor at each sampling time and tracking them backward for 30 days over the domain of 0° – 60° N, 70° E– 150° E, with resolution of $0.1^{\circ} \times 0.1^{\circ}$.

2.6 Fuel-specific fractions of C_{ff} by Keeling plot and Bayesian mixing model

The method to determine coal, oil, and natural gas (i.e., fossil fuel type) fractions of C_{ff} is described briefly using a Keeling plot (Miller and Tans, 2003) and the Bayesian mixing model (MixSIAR) (Stock et al., 2018). We calculated the excess $\delta(^{13}\text{C})$ (intercepts δ_{xs} , Eq. (4)) above the background level based on the best-fit lines in the Keeling plot. To determine the $\delta(^{13}\text{C})$ of the fossil fuel source (δ_{ff} , Eq. (5)), we estimated the weighted averages of the fossil fractions F_{ff} using a two end-member mixing analysis on C_{xs} . The $\delta(^{13}\text{C})$ of the biogenic source (δ_{bio}) was set to -26.1 ‰, which is the average $\delta(^{13}\text{C})$ value of the background air plus the -16.8 ‰ discrimination by the terrestrial ecosystem (Bakwin et al., 1998). We then estimated the coal, oil, and natural gas fractions of C_{ff} (F_{coal} , F_{oil} , and F_{ng} , Eqs. (6) and (7)) using a Bayesian tracer mixing model framework implemented as an open-source R package. The model used the δ_{ff} values as mixing data and the end-member $\delta(^{13}\text{C})$ signatures of coal, oil, and natural gas as the source data.

We adopted the end-member $\delta(^{13}\text{C})$ signatures measured in Beijing: $\delta_{coal} = (-24.3 \pm 0.2)$ ‰, $\delta_{oil} = (-28.9 \pm 0.5)$ ‰ and $\delta_{ng} = (-33.2 \pm 0.9)$ ‰ (Wang et al., 2022a). This selection was based on three considerations: First, coal $\delta(^{13}\text{C})$ signatures exhibit remarkable regional stability in China. Second, oil signatures from the Pearl River Mouth Basin of (-28.1 ± 1.6) ‰ (Cheng et al., 2013) show close agreement with Beijing values of (-28.9 ± 0.5) ‰. Third, measured natural gas signatures

like (-33.2 ± 0.9) ‰ in Beijing and (-32.0 ± 0.1) ‰ in Xi'an are significantly enriched compared to literature averages $[-39.5$ ‰ in Beijing and (-38.9 ± 2.6) ‰ in Pearl River Mouth Basin] (Ping et al., 2018; Quan et al., 2018), as using the lower literature values would lead to underestimation of natural gas contributions.

$$\delta_{\text{obs}} = C_{\text{bg}}(\delta_{\text{bg}} - \delta_{\text{xs}}) \times \frac{1}{C_{\text{obs}}} + \delta_{\text{xs}} \quad (4)$$

$$\delta_{\text{xs}} = F_{\text{ff}}\delta_{\text{ff}} + (1 - F_{\text{ff}})\delta_{\text{bio}} \quad (5)$$

$$\delta_{\text{ff}} = F_{\text{coal}}\delta_{\text{coal}} + F_{\text{oil}}\delta_{\text{oil}} + F_{\text{ng}}\delta_{\text{ng}} \quad (6)$$

$$1 = F_{\text{coal}} + F_{\text{oil}} + F_{\text{ng}} \quad (7)$$

2.7 Correlation of C_{ff} and CO and derivation of the emission ratio

We calculated Pearson correlation coefficient (r) between C_{ff} and CO enhancement ($\Delta\text{CO} = \text{CO}_{\text{obs}} - \text{CO}_{\text{bg}}$), and observational emission ratio of ΔCO to C_{ff} ($R_{\text{CO}/\text{CO}_{2\text{ff}}}$) [ppb ppm⁻¹ (nmol μmol⁻¹; similar hereafter)] using linear least squares regression. The $R_{\text{CO}/\text{CO}_{2\text{ff}}}$ ratios were derived from the regression slopes of ΔCO versus C_{ff} concentrations.

To correct for the contribution of non-fossil CO₂ in the observed enhancement, the emission ratio $R_{\text{CO}/\text{CO}_{2\text{ff}}}$ was estimated by dividing observed $R_{\text{CO}/\text{CO}_{2\text{ff}}}$ by 0.8. Previous $\Delta^{14}\text{CO}_2$ and CO-CO₂ studies (Turnbull et al., 2011; Lopez et al., 2013; Newman et al., 2016; Miller et al., 2020) have shown that ~10–30 % of the total CO₂ enhancement above background during daytime is typically of non-fossil origin, while CO is emitted almost exclusively from fossil-fuel combustion. Thus, the 20 % correction represents a reasonable first-order approximation for well-mixed afternoon conditions.

For comparison, the inventory emission ratio of CO to C_{ff} ($I_{\text{CO}/\text{CO}_{2\text{ff}}}$) [ppb ppm⁻¹ (nmol μmol⁻¹)] was calculated following as Lee et al. (2020) as:

$$I_{\text{CO}/\text{CO}_{2\text{ff}}} = E_{\text{CO}}/E_{\text{CO}_{2\text{ff}}} \times M_{\text{CO}_2}/M_{\text{CO}} \quad (8)$$

where E_{CO} and $E_{\text{CO}_{2\text{ff}}}$ represent the total CO and C_{ff} emissions, ~~respectively, in teragrams per year (Tg a⁻¹), summed over all grid cells within the relevant administrative boundaries from MEIC v1.4, MIX v2, and EDGAR 2024 inventories from the bottom-up national and urban inventory~~; and M_X refers to the molar masses of CO and CO₂ in grams per mole (g mol⁻¹).

3 Results and discussions

3.1 Background selection

We conducted atmospheric observations of CO₂ and its carbon isotope composition ($\Delta^{14}\text{C}$) and $\delta^{13}\text{C}$) in Guangzhou, Shenzhen, Zhanjiang, and Shaoguan in Guangdong Province, South China, during the summer and winter of 2022. To attribute CO₂ enhancements (C_{xs}) to a particular region, it is necessary to isolate the component of the observed concentration attributable to fluxes within the region by removing the background (Karion et al., 2021). High-elevation

mountains, representing the free troposphere, were considered ideal background locations for use in this study (Turnbull et al., 2009). Specifically, the Nanling site (NL, 1700 m above sea level (a.s.l.)), one of the 30 sampling sites of this study (SG5; Table A1), was selected because it serves as the nearest regional background site for the study areas with relatively complex boundary conditions (for more reasons see Appendix D). The “annual” CO₂ and $\Delta(^{14}\text{C})$ averages at NL station, calculated as averages of summer and winter measurements, were $(418.5 \pm 7.3) \mu\text{mol mol}^{-1}$ and $(-7.1 \pm 3.9) \text{‰}$, respectively. These values closely match those observed at Jungfraujoch (JFJ, 3580 m a.s.l.) and appear in the upper-right section of the Keeling plot of $\Delta(^{14}\text{C})$ and CO₂ (i.e., scatter plot between $\Delta(^{14}\text{C})$ and inverse of CO₂ mole fractions) representing background conditions (Pataki et al., 2003). This positioning becomes evident when comparing with Waliguan (WLG, 3890 m a.s.l.) station data (Fig. 2). The advantage of using the Keeling plot method to screen background data is that it simultaneously accounts for both higher values of $\Delta(^{14}\text{C})$ and lower values of CO₂ (Zhou et al., 2024). The $\Delta(^{14}\text{C})$ averages at NL were the highest among the 30 sampling sites considered in this study, with values of $(-3.7 \pm 1.3) \text{‰}$ in summer and $(-10.6 \pm 0.8) \text{‰}$ in winter (Table A1). We selected a regional (rather than urban) background site in 2022 to capture higher C_{ff} concentrations. This approach establishes a lower baseline for analyzing C_{ff} reduction in subsequent sections.

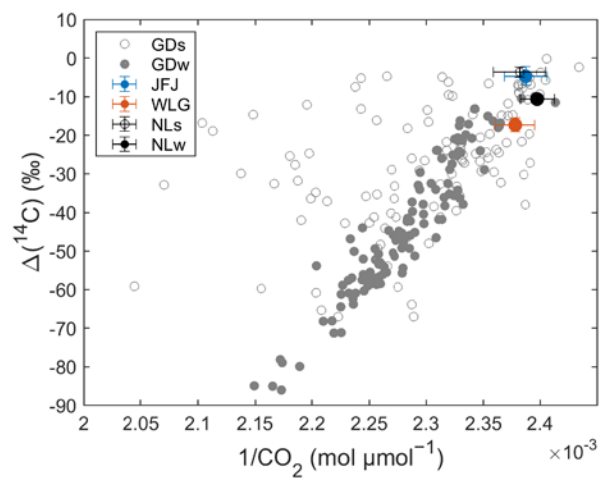
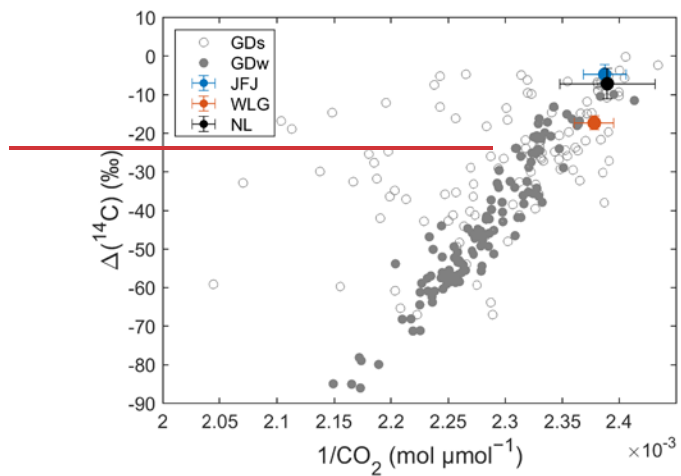


Figure 2: Keeling plot of CO_2 and $\Delta(^{14}\text{C})$ measurements from Guangdong Province in summer (GDs) and winter (GDw), and background stations including JFJ (Jungfrauoch) (Emmenegger et al., 2024b, a), WLG (Waliguan) (Liu et al., 2024; Lan et al., 2023), and NL (Nanling, this study) in 2022. For the JFJ background site, the complete 2022 dataset was used to calculate a true annual mean. For the WLG background site, the CO_2 concentrations at the WLG station were obtained from the World Data Centre for Greenhouse Gases (WDCGG, <https://gaw.kishou.go.jp/>, last accessed: April 21, 2024), while $\Delta(^{14}\text{C})$ observations were

obtained from Liu et al. (2024). For the NL background site, CO₂ and $\Delta(^{14}\text{C})$ observations were obtained from two campaigns in August and December 2022, representing typical summer and winter conditions.

270 3.2 CO₂, $\Delta(^{14}\text{C})$, C_{xs} and C_{ff} concentrations

CO₂ concentrations in Guangzhou, Shenzhen, Zhanjiang, and Shaoguan were (438.8 ± 12.3), (435.0 ± 12.7), (444.2 ± 17.2), and (431.6 ± 10.5) μmol mol⁻¹ (multisite mean and one-sigma standard deviation), respectively; the corresponding $\Delta(^{14}\text{C})$ values were (-40.7 ± 13.4) ‰, (-37.2 ± 24.1) ‰, (-28.8 ± 13.8) ‰, and (-25.0 ± 14.9) ‰, respectively. Relative to the background, CO₂ concentrations in the four cities were enhanced by (20.3 ± 12.5), (16.5 ± 13.5), (25.8 ± 16.7), and (13.1 ± 10.1) μmol mol⁻¹ (C_{xs}), respectively; the mean $\Delta(^{14}\text{C})$ was depleted by (-33.6 ± 11.3) ‰, (-29.9 ± 22.3) ‰, (-21.5 ± 11.7) ‰, and (-17.8 ± 15.7) ‰ ($\Delta\Delta(^{14}\text{C})$), respectively, reflecting the marked influence of ¹⁴C-free CO₂ emissions from fossil fuel combustion. The fossil fuel and biogenic fractions of C_{xs}, C_{ff} and C_{bio}, were determined using a two end-member mixing analysis. The C_{ff} fractions were (79 ± 5) %, (73 ± 6) %, (59 ± 4) %, and (53 ± 13) % during winter in Guangzhou, Shenzhen, Zhanjiang, and Shaoguan, respectively. In comparison with other cities worldwide (Table E1, Fig. E1), we observed higher C_{ff} fractions (>70 %) in some megacities and supercities compared with large and medium-sized cities. Noting that the C_{ff}/C_{xs} ratio is critically sensitive to background selection. Regional backgrounds (as implemented here) introduce C_{bio} contributions from surrounding rural/ agricultural sources to C_{xs}, whereas local urban backgrounds effectively isolate urban emissions by filtering out these external biogenic signals, thereby increasing the apparent C_{ff} fraction compared to regional background approaches. The consistent adoption of regional background methodologies across all studies in Table E1 ensures the comparative validity of the results, as they share a common framework for accounting for C_{bio} influences from peripheral non-urban sources. The derived annual C_{ff} averages are (15.3 ± 5.2), (13.7 ± 10.2), (10.0 ± 5.2), and (8.2 ± 7.0) μmol mol⁻¹ in Guangzhou, Shenzhen, Zhanjiang, and Shaoguan, respectively, based on the mass balance equations of CO₂ and $\Delta(^{14}\text{C})$. These C_{ff} concentrations were low to moderate compared with those in other cities globally (Table E2, Fig. E1), despite the high emissions in Guangzhou and Shenzhen from inventories (Fig. 1).

290

3.3 Spatial distribution and seasonal variations

The spatial differences observed in C_{ff} primarily reflect the combined influence of emission intensity and atmospheric transport rather than direct emission magnitudes. We first identified potential source regions that are likely to contribute to the observed ~~of~~ C_{ff} variability by analyzing its spatial distribution and seasonal variations. Higher C_{ff} levels were typically observed at ~~is found to be emitted dominantly from~~ densely populated urban-downtown sites/areas (GZ6 and GZ5; SG3 and SG2) in Guangzhou during summer (GZs) and Shaoguan during winter (SGw), forming an “urban C_{ff} dome” (Fig. 1c). This was further supported by significant positive correlations between the C_{ff} measurements and the corresponding 1x1 km gridded ODIAC (Oda and Maksyutov, 2011; Oda and Maksyutov, 2024) inventory emissions in GZs ($r = 0.53$, $p = 0.1$) and

SGw ($r = 0.91, p = 0.03$). The “urban C_{ff} dome” indicates that C_{ff} is mainly derived from the localized fossil fuel combustion, which is likely to be influenced by the urban topography. That is, downtown Guangzhou and downtown Shaoguan are surrounded by mountains to the east, north, and west. In contrast, we found that ~~higher C_{ff} was mainly emitted~~ from western industrial areas ~~and airport~~ (SZ2) in Shenzhen during summer (SZs), and from port areas (ZJ5>ZJ2; ZJ2>ZJ3>ZJ4) in Zhanjiang during winter (ZJw) and summer (ZJs, by atmospheric transport) (Fig. 1ehg).

Atmospheric transmission of C_{ff} from potential source regions was observed at large spatial scales combined with air mass back trajectories by the Hybrid Single-Particle Lagrangian Integrated Trajectory (HYSPLIT) model (Stein et al., 2015) and emission footprints by the FLEXPART dispersion model (Pisso et al., 2019). Shaoguan exhibited higher C_{ff} concentrations in summer ($(10.7 \pm 8.3) \mu\text{mol mol}^{-1}$) than in winter ($(5.8 \pm 4.4) \mu\text{mol mol}^{-1}$). This was likely attributable to atmospheric trapping of emissions from the Pearl River Delta (PRD) urban agglomeration, as illustrated by air mass back trajectories (HYSPLIT, Fig. F1a) and emission footprints (FLEXPART, Fig. 3g), rather than high local emissions due to lower summer emissions from inventories. In contrast, we found higher C_{ff} concentrations in winter compared with those in summer in Guangzhou ($(17.7 \pm 3.5) > (12.9 \pm 5.6) \mu\text{mol mol}^{-1}$), Shenzhen ($(18.0 \pm 9.9) > (9.2 \pm 8.5) \mu\text{mol mol}^{-1}$), and Zhanjiang ($(12.1 \pm 5.1) > (7.6 \pm 4.3) \mu\text{mol mol}^{-1}$), consistent with the values in 14 other Chinese cities (Zhou et al., 2020). The higher winter concentrations found in Guangzhou, Shenzhen, and Zhanjiang in this study likely resulted from atmospheric trapping of emissions in the shallow planetary boundary layer, and high local emissions, because ODIAC (MEIC) indicates that winter emissions were 8 %, 10 %, and 11 % (17 %, 22 %, and 14 %) higher, respectively, than those in summer (Oda and Maksyutov, 2024; Meic, 2023). The atmospheric trapping of emissions is higher than local emissions during winter in Guangzhou (GZw) and Shenzhen (SZw), which is supported by higher C_{ff} concentrations occurring in downwind areas (GZ2, GZ6, and GZ10; SZ3 and SZ4) compared with upwind areas (GZ1 and GZ3; SZ8 and SZ9). The air mass back trajectories (HYSPLIT, Fig. F1b) and emission footprints (FLEXPART, Fig. 3bd) showed that the major source region was traced to the Yangtze River Delta (YRD) urban agglomeration in East China, and a portion from North China via long-range transport (Fig. F2ef). The major source region from the YRD was also reported in a study of CFC-11 in Shenzhen (Chen et al., 2024).

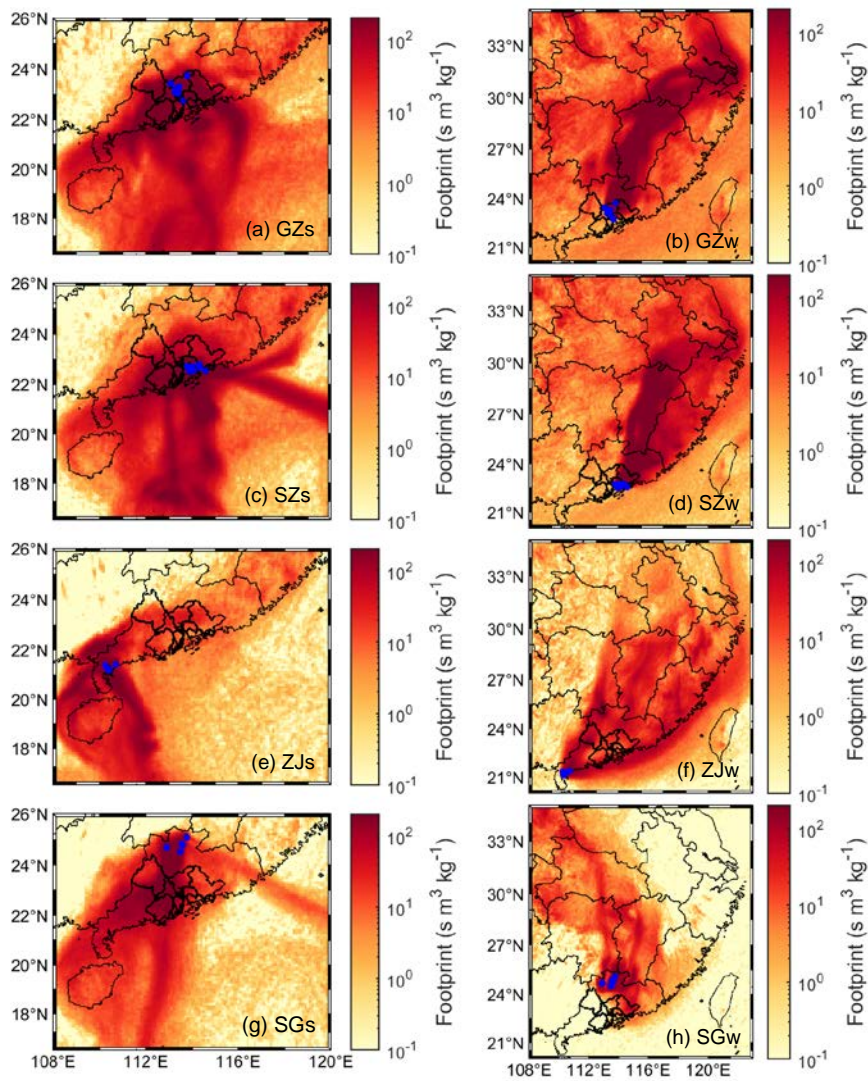


Figure 3: FLEXPART footprints simulating C_n emissions in summer (s) and winter (w) for (a and b, GZ) Guangzhou, (c and d, SZ) Shenzhen, (e and f, ZJ) Zhanjiang, and (g and h, SG) Shaoguan at heights from 0–100 m a.s.l. over a period of 30 days. **Blue**

squares are shown as enlarged maps in the right figures. Blue points represent the locations of sampling sites. Black lines indicate the boundaries of continents (left), Chinese provinces (left, bold), and the nine cities of the PRD (right, bold) taken from Natural Earth (<https://www.naturalearthdata.com/>, last accessed: 9 March 2024).

3.4 Historical variations

3.4.1 Meteorological typicality of the sampling months

As shown in Fig. G1, all five meteorological variables (10 m eastward wind, U10; 10 m northward wind, V10; 2 m air temperature, T2M; surface pressure, SP; and planetary boundary-layer height, PBLH) at all Guangzhou sites exhibit $|z| < 1$, indicating that both August and December 2022 were meteorologically typical relative to the same-year seasonal background and the 2010–2021 climatological baselines. August 2022 featured slightly weaker easterly winds and near-climatological boundary-layer heights, while December 2022 was characterized by prevailing northerly flow and typical boundary-layer ventilation.

Complementary ERA5 wind-rose analyses (Fig. G2) and 72 h HYSPLIT back-trajectory simulations (Fig. F1) confirm that both months followed the canonical East Asian monsoon regimes—maritime inflow during summer and continental outflow during winter. Using GZ7 as an illustrative example representative of central Guangzhou, the ERA5 wind roses show dominant east–east-southeasterly ($90\text{--}135^\circ$) winds in August 2022, typically $3\text{--}8\text{ m s}^{-1}$. In comparison, JJA 2022 and the 2010–2021 JJA climatology peak in the south–south-westerly sector ($157.5\text{--}225^\circ$), representing a within-sector rotation ($90\text{--}225^\circ$) rather than a regime change. ERA5 anomalies of U10, V10, and PBLH remain below 1σ , confirming transport typicality. HYSPLIT trajectories indicate that August 2022 air masses primarily originated over the South China Sea, consistent with summer maritime inflow. For December 2022, the ERA5 wind roses display a clear north–northeasterly ($0\text{--}45^\circ$) dominance with $3\text{--}8\text{ m s}^{-1}$ speeds. The DJF 2022 composite and the 2010–2021 DJF climatology show nearly identical northerly continental patterns, typical of the East Asian winter monsoon. HYSPLIT back trajectories confirm that the air parcels predominantly arrived from northern continental China under prevailing northerlies. These results demonstrate that both sampling periods were representative of their respective seasonal transport conditions. Consequently, atmospheric transport variability is unlikely to bias the reported C_{ff} trend or the inferred changes in $R_{CO_2/CO_{2ff}}$ ratios and fuel-type contributions.

3.4.2 Representativeness of weekly flask samples

Each flask represents approximately 15–20 min of integrated air, and about 40 samples were collected per month across ten stations, providing broad spatial and temporal coverage. To evaluate how representative these discrete samples are for the respective seasons, we compared ERA5 diagnostics (PBLH, wind speed, and wind direction) during sampling days with the corresponding monthly means. The results show that meteorological conditions during sampling closely matched monthly climatological averages, confirming that no unusual stagnation or transport anomalies occurred on the sampling days.

360 ERA5 wind roses (Fig. G2) and HYSPLIT 72-h back-trajectories (Fig. F1) further confirm that the flask collection periods coincided with the prevailing summer (90–225°) and winter (0–45°) monsoon sectors. Hence, the samples captured the dominant seasonal transport regimes rather than isolated short-term events. This demonstrates that the weekly flask observations are meteorologically and dynamically representative of their respective seasonal backgrounds, minimizing the potential bias from short-term synoptic variability. Consequently, the derived C_{ff} concentrations from these samples can be regarded as seasonally robust, and the subsequent interannual comparisons mainly reflect emission-driven rather than sampling-driven differences.

3.4.3 Historical variation of C_{ff} concentrations

365 Because both August and December 2022 were meteorologically typical, the observed inter-annual differences in C_{ff} are attributed mainly to emission rather than transport variability. To ensure comparability, all available historical datasets (Table H1) were harmonized to identical sites, seasons, and local-time windows, and recalculated using unified background references (Table H2, Fig. 4a). These harmonized datasets minimize transport and spatial biases, allowing the remaining differences in C_{ff} mole fractions to be interpreted as primarily emission driven.

370 For Guangzhou, a site-specific long-term comparison was conducted at the GZ7 urban station, which was also used by Ding et al. (2013). In their study, C_{ff} was derived from flask observations collected around 20:00 LT (post-rush-hour) using a $\Delta(^{14}C)$ background based on corn-leaf samples from Qinghai, Gansu, and Tibet. Such a background likely represents a different air-mass domain from Guangzhou. In contrast, the present study used atmospheric $\Delta(^{14}C)$ observations from the NL regional background site, which directly samples the same regional air masses influencing Guangzhou. To make the datasets directly comparable, the winter 2010 C_{ff} values from Ding et al. (2013) and the winter 2022 values from this work were recalculated using the NL tree-ring $\Delta(^{14}C)$ record (Li et al., 2025b) as a common background reference. This adjustment yields $27.3 \pm 16.9 \mu\text{mol mol}^{-1}$ for 2010 and $11.6 \pm 3.4 \mu\text{mol mol}^{-1}$ for 2022, indicating a pronounced reduction in fossil-fuel-derived CO_2 over the decade.

375 Because sampling times differ (20:00 vs 14:00 LT), we quantified the expected diurnal C_{ff} contrast using continuous CO observations near GZ7. ΔCO increased from 168 ppb at 14:00 to 221 ppb at 20:00, corresponding to a 21 % C_{ff} nighttime enhancement (Scheme 1, Appendix H1). A supplementary analysis using the winter 2023–2024 dataset gave a 35 % enhancement (Scheme 2, Appendix H1). These findings suggest that the evening C_{ff} level is typically 21–35 % higher than the well-mixed afternoon value due to weaker nocturnal boundary-layer mixing. Applying this correction, the 2010 nighttime C_{ff} ($27.3 \pm 16.9 \mu\text{mol mol}^{-1}$) corresponds to an afternoon-equivalent concentration of $\approx 17.7\text{--}21.6 \mu\text{mol mol}^{-1}$, which remains substantially higher than the 2022 value of $11.6 \pm 3.4 \mu\text{mol mol}^{-1}$.

380 Even after harmonizing the background and correcting for sampling-time bias, C_{ff} in Guangzhou declined by $\approx 34\text{--}46$ % between 2010 and 2022, confirming a genuine and statistically significant ($p < 0.01$) decrease in fossil-fuel CO_2 . FLEXPART footprint analyses for 2010 and 2022 show similar source-sensitivity patterns centered on the Guangzhou urban core, confirming that GZ7 remains spatially representative of Guangzhou's urban domain.

390 Comparable harmonized analyses were performed for other Chinese cities (Tables H1 and H2; Fig. 4a). For Beijing, all
measurements originate from the urban rooftop site of the Research Center for Eco-Environmental Sciences, Chinese
Academy of Sciences (RCEES). The $\Delta(^{14}\text{C})$ background used in Zhou et al. (2020) was based on Qixianling Mountain
(QXL), whereas Wang et al. (2022b) adopted the Waliguan (WLG) background. All C_{ff} values were recalculated using WLG
as a common reference background with the 2015 value from Niu et al. (2016). After this correction, the 2014–2016 winter
395 C_{ff} value increases slightly from $27.0 \pm 0.3 \mu\text{mol mol}^{-1}$ to $27.6 \pm 0.3 \mu\text{mol mol}^{-1}$, ensuring consistency across datasets.
Relative to this harmonized baseline, the subsequent decline to $19.7 \pm 22.0 \mu\text{mol mol}^{-1}$ by winter 2020 (Wang et al., 2022b)
represents an approximate 29 % reduction ($p < 0.05$). This trend is consistent with regional fossil-fuel CO_2 emission
reductions and corroborated by independent $\Delta(^{14}\text{C})$ tree-ring records showing a peak near 2010 in Beijing (Niu et al., 2024).
For Xi'an, at the Institute of Earth Environment, Chinese Academy of Sciences (IEECAS) urban site, C_{ff} fell by 36 % from
400 $(40.1 \pm 3.8) \mu\text{mol mol}^{-1}$ in 2011–2013 to $(25.7 \pm 1.1) \mu\text{mol mol}^{-1}$ in 2014–2016 ($p < 0.001$) (Zhou et al., 2022). Suburban
sites declined by ≈ 12 % from $(23.5 \pm 6.5) \mu\text{mol mol}^{-1}$ in 2016 (Wang et al., 2018) to $(13.1 \pm 10.9) \mu\text{mol mol}^{-1}$ in 2021–
2022 (Liu et al., 2024) ($p < 0.05$). These decreases are consistent with independent $\Delta(^{14}\text{C})$ tree-ring records indicating
emission peak near 2013 in Xi'an (Niu et al., 2024).
Overall, the harmonized, site-specific, and time-of-day-corrected comparisons demonstrate statistically significant reductions
405 in fossil-fuel CO_2 across China's major urban centers. For Guangzhou particularly, the combined evidence—consistent
background domain, typical meteorology, verified sampling representativeness, and quantified diurnal correction—provides
strong support that the observed C_{ff} decline reflects genuine decarbonization rather than artifacts of sampling or transport
variability. We observed a 35 % decline in C_{ff} concentrations in Guangzhou from $(23.7 \pm 12.9) \mu\text{mol mol}^{-1}$ in 2010–2011
(Ding et al., 2013) to $(15.3 \pm 5.2) \mu\text{mol mol}^{-1}$ in 2022 in this study ($p < 0.01$, two-tailed t test) (Fig. 4a, Table E2). To
410 conservatively validate this reduction, we employed a dual-method analytical framework for 2022 data designed to
maximize C_{ff} estimates: (1) adopting a regional (instead of urban) background to establish a lower baseline, and (2) applying
both literature-derived and simulation-based disequilibrium corrections (β) to address potential biases. This validation
confirmed 2022 C_{ff} values remain statistically significantly lower than 2010–2011 levels across all the observational data ($p <$
0.01). Additional site-specific validation at GZ7 station showed a 41 % winter reduction from $(28.4 \pm 15.0) \mu\text{mol mol}^{-1}$ in
415 2010 to $(16.8 \pm 3.4) \mu\text{mol mol}^{-1}$ in 2022, with equivalent statistical significance ($p < 0.01$). By adjusting baseline
assumptions and correction parameters, this methodology ensures that the reported decline is robust against potential
overestimations in 2022. It provides reliable evidence for Guangzhou's decarbonization progress, even under maximized C_{ff}
calculation scenarios in 2022. Furthermore, this observed decline in C_{ff} is consistent with reported emission reductions in
major source regions of South and East China (e.g., Hebei, Shandong, Zhejiang, and Guangdong; Fig. F2) according to the
420 MEIC inventory
We found that reductions in C_{ff} concentration also occurred in other Chinese cities, and some cities outside China.
Statistically significant reductions were observed in Chinese megacities and supercities (Fig. 4a, Table E2): Beijing showed
a 32 % decrease from $(39.7 \pm 36.1) \mu\text{mol mol}^{-1}$ in 2014 to $(27.0 \pm 0.3) \mu\text{mol mol}^{-1}$ in 2014–2016 ($p < 0.05$, two-tailed t test)

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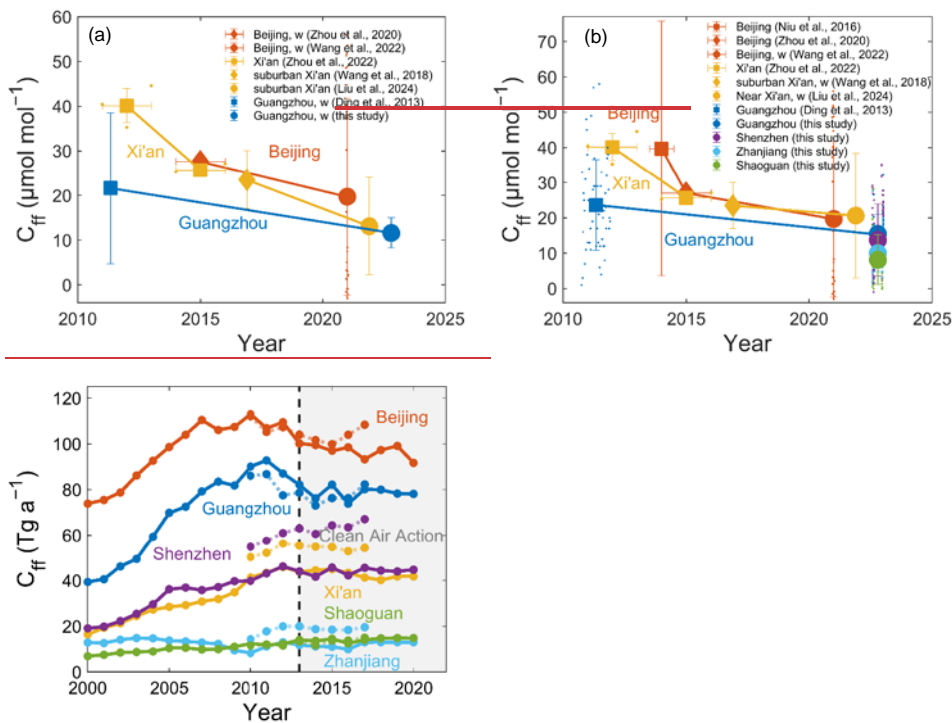
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425 and a 50 % decrease to $(19.7 \pm 22.0) \mu\text{mol mol}^{-1}$ by winter 2020 ($p < 0.01$); Xi'an decreased by 36 % from (40.1 ± 3.8)
 $\mu\text{mol mol}^{-1}$ in 2011–2013 to $(25.7 \pm 1.1) \mu\text{mol mol}^{-1}$ in 2014–2016 ($p < 0.001$), with suburban areas declining by 12 % from
 $(23.5 \pm 6.5) \mu\text{mol mol}^{-1}$ in 2016 to $(13.1 \pm 10.9) \mu\text{mol mol}^{-1}$ in 2021 ($p < 0.05$). These trends are consistent with tree-ring
 $\Delta(^{14}\text{C})$ records showing peaks in urban C_{ff} emissions for Beijing (2010) and Xi'an (2013) (Niu et al., 2024). Globally,
Krakow, Poland showed a 93 % decrease from $27.5 \mu\text{mol mol}^{-1}$ in 1989 to $1.98\text{--}2.18 \mu\text{mol mol}^{-1}$ in 2005–2009 ($p < 0.001$);
430 Heidelberg, Germany a 2 % decrease from $(11.09 \pm 0.24) \mu\text{mol mol}^{-1}$ in 1986–1996 to $(10.92 \pm 0.34) \mu\text{mol mol}^{-1}$ in 1997–
2007 ($p = 0.04$), and Los Angeles, USA a 42 % decrease from $(22.9 \pm 5.6) \mu\text{mol mol}^{-1}$ in 2006–2013 to $(13.2 \pm 9.4) \mu\text{mol}$
 mol^{-1} in 2014–2016 ($p < 0.01$) (Table E2, Fig. E1). The consistent statistical significance ($p < 0.05$ threshold unless noted)
across atmospheric measurements and proxy records underscores the effectiveness of emission mitigation policies in these
cities.

435 We found that C_{ff} concentration reduction did not only occur in the above-mentioned cities, but also probably in their source
regions. The winter C_{ff} emission source regions of Guangzhou and Shenzhen are highly overlapped with China's high C_{ff}
emitting areas: North China, the YRD in East China, and the PRD in South China (Figs. 3 and F2). These areas are also
mostly consistent with the provinces that attained the higher C_{ff} emission reductions due to stricter pollution control policies.
These provinces include Hebei, Shandong, Zhejiang, Shanxi, and Henan in North and East China according to the MEIC
inventory (Shi et al., 2022), supporting the interpretation of a widespread decarbonization trend. Therefore, it's very likely
440 that C_{ff} emissions are also decreasing or in a declining growth rate in China's high C_{ff} emitting areas.

Similar reductions were found in C_{ff} emissions from 2012 to 2020 according to the MEIC inventory (Shi et al., 2022), such
as Guangzhou (by 16 % from 2011), Shenzhen (by 3 %), Zhanjiang (by 0.1 %), Beijing (by 16 %), and Xi'an (by 9 %) (Fig.
4b), particularly in the industrial and power sectors (Li et al., 2017). We also found such declines in the MIXv2 Asian
emission inventory (MIXv2, excluding Shenzhen and Shaoguan) (Li et al., 2024) and another carbon inventory for most
445 Chinese cities (Zhang et al., 2024), but not in the ODIAC (Oda and Maksyutov, 2024) and the Emissions Database for
Global Atmospheric Research (EDGAR) (Crippa et al., 2023). In fact, the mitigation of C_{ff} emissions in China's MEIC
inventory was primarily driven by heterogeneous trends across cities: 38 % exhibited sustained emission reductions, 29 %
showed an initial decline followed by a rebound, while 33 % maintained increasing trajectories. Notably, cities achieving
sustained reductions were disproportionately concentrated in larger cities, comprising 86 % of megacities, 43 % of
450 supercities, and 43 % of Type I large cities (populations of 3–5 million). In contrast, smaller cities showed lower mitigation
prevalence, with only 34 % of Type II large cities (1–3 million) and 38 % of medium/ small cities attaining emission
decreases.



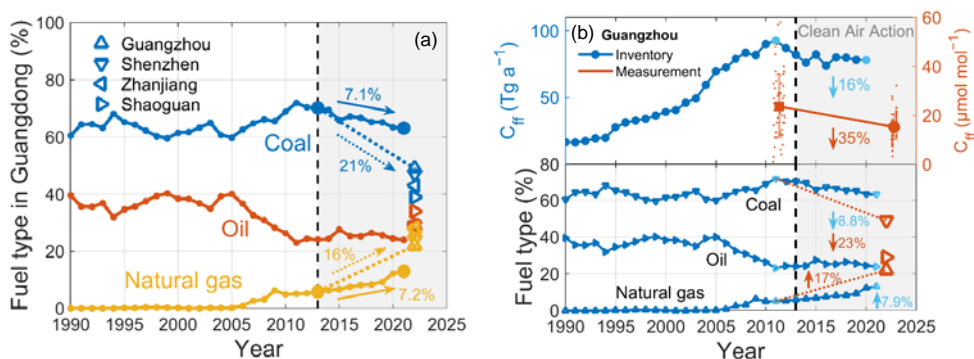
455 **Figure 4:** (a) Harmonized comparison of C_{fr} mole fractions at identical sites, seasons, and sampling times, after correction to
common background conditions. C_{fr} concentrations from atmospheric measurements (Wang et al., 2022b; Zhou et al., 2022; Ding
460 et al., 2013; Zhou et al., 2020; Wang et al., 2018) in Beijing, Xi'an, and Guangzhou, Shenzhen, Zhanjiang, and Shaoguan. The
large symbols indicate annual means, multiyear averages, or winter means (w) of corrected C_{fr} in Table H2, and the small symbols
represent the corresponding scattered measurements. C_{fr} observations are calculated as enhancements over background Nanling
for Guangzhou and Waliguan for Beijing and Xi'an. The C_{fr} concentrations in winter in Beijing obtained from Wang et al.⁴⁰ were
estimated based on the background $\Delta(^{14}\text{CO}_2)$ measurements from Waliguan (Liu et al., 2024). The y-axis error bars indicate
465 uncertainty, and the x-axis error bars represent the observed period. (b) C_{fr} emissions from the MEIC (solid lines) (Li et al., 2017;
Meic, 2023; Zheng et al., 2018) and MIXv2 (dotted lines) (Li et al., 2024) inventories in Beijing, Xi'an, Guangzhou, Shenzhen,
Zhanjiang, and Shaoguan since 2010. The vertical dashed line indicates the year 2013 when China's Clean Air Action Plan
was implemented.

3.5 Driver factors

3.5.1 Coal-to-gas transition

470 We first determined the coal, oil, and natural gas fractions of C_{ff} using the Keeling plot of $\delta(^{13}C)$ and CO_2 (i.e., scatter plot
between $\delta(^{13}C)$ and inverse of CO_2 mole fractions) and the Bayesian mixing model (MixSIAR) (Stock et al., 2018) during
winter 2022. The fractions in winter were $(49 \pm 25) \%$, $(29 \pm 22) \%$, and $(22 \pm 19) \%$, respectively, for Guangzhou, $(47 \pm$
 $25) \%$, $(29 \pm 21) \%$, and $(24 \pm 20) \%$ for Shenzhen, $(43 \pm 24) \%$, $(29 \pm 21) \%$, and $(28 \pm 21) \%$ for Zhanjiang, and $(39 \pm$
475 $24) \%$, $(34 \pm 23) \%$, and $(27 \pm 21) \%$ for Shaoguan. Coal combustion was the largest contributor to C_{ff} emissions, followed in
descending order by oil combustion and natural gas combustion. Compared with other cities around the world (Table IG1),
we found natural gas was the primary fuel type consumed in Paris (70 %) (Lopez et al., 2013) and Beijing $[(55 \pm 9) \%$
(Wang et al., 2022b), whereas oil was the main fuel type consumed in Los Angeles ($>50 \%$) (Djuricin et al., 2010; Newman
et al., 2016). Coal remains the primary fossil fuel used in Xi'an $[(72.6 \pm 10.4) \%$ in 2014 and $(54 \pm 4) \%$ in 2019) (Wang et
al., 2022b; Zhou et al., 2014), Guangzhou (49 % in 2022), and Shenzhen (47 % in 2022). Notably, cities with high C_{ff}
480 emissions consume all three types of fossil fuels, with the dominant fuel type varying by city. Coal remains the primary
fossil fuel used in many Chinese cities.

The reduction in C_{ff} concentrations can be attributed to changes in energy systems as a result of China's clean air measures
(Shi et al., 2022). A major contribution has been the reduction in coal usage and the shift to low-carbon energy sources such
as natural gas. During 2013–2022, the share of coal in the energy mix decreased by 4.9 % in China and by 7.1 % in
485 Guangdong Province, whereas the share of natural gas increased by 3.0 % in China and by 7.2 % in Guangdong Province,
according to the MEIC inventory (Li et al., 2017; Zheng et al., 2018; Meic, 2023; Xu et al., 2024). By applying the coal, oil,
and natural gas fractions of C_{ff} derived from our measurements, it's likely that coal usage in Guangdong Province since 2013
have decreased $\geq 21 \%$, and natural gas usage have increased by $\geq 16 \%$ (Fig. 5a). Similarly, in Guangzhou city, it's likely
that coal usage since 2011 has decreased by 23 % instead of by 8.8 % (Fig. 5b), and natural gas usage has increased by 17 %
490 instead of by 7.9 %, assuming that the fuel type fractions of C_{ff} in Guangzhou city were the same as those in Guangdong
Province in the inventory.



495 Figure 5: (a) Coal, oil, and natural gas fractions of C_{ff} in Guangdong Province from the MEIC inventory from 1990 to 2021 (points), and in the cities of Guangzhou, Shenzhen, Zhanjiang, and Shaoguan from measurements in this study in 2022 (triangles).
 (b) (Top) Comparison of reductions in C_{ff} inventory emissions (blue) and measured C_{ff} concentrations (red) resulting from
 (Bottom) reduced coal usage and increased natural gas usage in Guangzhou. The vertical dashed line indicates the year 2013 when
 China's Clean Air Action Plan was implemented.

500 3.5.2 Combustion efficiency improvement

We first calculated $R_{CO/CO_{2ff}}$ ratios at each measurement site and found higher ratios in summer than winter (Fig. JH1). However, we focused only on observations in winter for four reasons. First, summer CO shows greater instability as its atmospheric lifetime depends on OH radical production, which is enhanced through photochemical reactions (e.g., CH_4 oxidation) under intense solar radiation, making CO a less reliable fossil fuel tracer (Rosendahl, 2022). Second, winter exhibits stronger $\Delta CO-C_{ff}$ correlations ($r > 0.6$, $p < 0.01$; Fig. JH2) with better regional representativeness due to extended CO atmospheric lifetime from slower CO oxidation rates. Third, the winter $\Delta CO-C_{ff}$ relationship better captures anthropogenic emission characteristics compared to other seasons. Fourth, weaker vertical mixing in winter accentuates local emission impacts (Wang et al., 2010). In addition, the lower C_{ff} signals observed in summer lead to higher uncertainty in the regression slope and thus greater uncertainty in the $R_{CO/CO_{2ff}}$ ratios, as also noted in Maier et al. (2024), which can be seen in the larger error bars of the summer data in Fig. J2.

We then estimated winter 2022 $R_{CO/CO_{2ff}}$ ratios across Chinese cities using $\Delta CO-C_{ff}$ regression slopes (Fig. JH2), with spatial variations primarily attributed to differences in fuel composition and combustion efficiency (Graven et al., 2009). CO is generated through incomplete combustion of both fossil fuels and biomass. These spatial patterns are consistent with combustion characteristics showing biomass burning produces higher CO emissions per unit energy than fossil fuel combustion (Akagi et al., 2011). As shown in Fig. H1, suburban/rural sites (GZ1, SZ9, ZJ1, SG1) exhibited significantly higher ratios than urban sites (GZ5, SZ7, ZJ4, SG3): $GZ1 > GZ5$ (30.4 ± 10.0) $>$ (19.8 ± 4.6) $nmol \mu mol^{-1}$, $SZ9 > SZ7$

((41.3 ± 23.0) > (14.8 ± 2.4) nmol μmol⁻¹), ZJ1 > ZJ4 ((41.2 ± 3.6) > (11.9 ± 6.4) nmol μmol⁻¹), and SG1 > SG3 ((26.7 ± 2.9) > (15.1 ± 3.6) nmol μmol⁻¹). This pattern is in agreement with previous studies attributing elevated ratios in non-urban areas to biomass burning contributions (Rosendahl, 2022). In contrast, megacities showed 35–40 % lower ratios (Guangzhou: 13.3 ± 3.1 nmol μmol⁻¹, Shenzhen: 13.5 ± 2.4 nmol μmol⁻¹) compared to smaller cities (Zhanjiang: 22.6 ± 5.0 nmol μmol⁻¹, Shaoguan: 21.7 ± 6.4 nmol μmol⁻¹; $p < 0.01$; Fig. JH2), suggest higher fossil fuel combustion efficiency and/or lower biomass burning inputs. Guangzhou's ratios are dominated by improved fossil fuel combustion efficiency due to having the highest biomass burning emissions among the four studied cities in the EDGAR2024 inventory, while Shenzhen's ratios are attributed to both factors with nearly negligible biomass contributions corresponding to its 2017 biomass boiler phase-out policy.

We retrieved historical R_{CO_2/CO_2ff} data from observations in China by estimation from $\Delta(^{14}C)$ measurements and correction from R_{CO_2/CO_2} (increased by 20 %) (Table JH1), and I_{CO_2/CO_2ff} data from the MEIC, MIXv2, and EDGAR inventories (Fig. 6). Given the minor contribution of biomass burning (BB)-related CO emissions across all inventories, with I_{CO_2/CO_2ff} ratio below 0.003 (MEIC, incorporating OBBEIC data (Song et al., 2009; Huang et al., 2012)), less than 1.0 (MIXv2), and declining from 3.8 (1990) to 1.1 (2022) in EDGAR, we assume that interannual variability in BB emissions has negligible influence on the overall emission ratios. Observational and inventory data show a sustained decline in China's R_{CO_2/CO_2ff} and I_{CO_2/CO_2ff} ratios over the past 30 years (Fig. 6a), demonstrating that efforts to improve fossil fuel combustion efficiency are effective (Wang et al., 2010; Lee et al., 2020), which is another factor contributing to the reduction in C_{ff} concentrations. The MEIC inventory attributes this trend to spatiotemporally heterogeneous mitigation pathways: 72 % of the cities started I_{CO_2/CO_2ff} reductions during 1990–1994, while the remaining 28 % (mainly concentrated in the western provinces) exhibited a delayed start until 1995–2004. The implementation of China's clean air policies since 2013 has systematically phased out small, inefficient combustion facilities and replaced them with centralized, high efficient, and clean energy infrastructure (Shi et al., 2022). The phase-out of coal-fired industrial boilers during 2013–2020 reduced CO₂ emissions by (1.5 ± 0.3) Gt, accounting for 12 % of the national industrial emission reduction (Li, 2023). These technological transitions enhanced combustion efficiency by >10 %, and reduced coal-dominated energy intensity by 40 % across the sector. The MEIC inventory showed that these synergistic measures resulted in significant energy savings, with a net reduction of 0.25 gigatonnes of coal equivalent (Gtce) in 2020 and a cumulative reduction of 1.06 Gtce over the policy implementation period (Shi et al., 2022). Critically, the efficiency-driven transition decoupled energy demand from C_{ff} emissions, with combustion optimization directly reducing coal consumption 1–2 % and C_{ff} emissions by 1–3 Gt per year after 2015 (Le Quéré et al., 2016; Friedlingstein et al., 2023b).

We systematically compared observational R_{CO_2/CO_2ff} values with inventory I_{CO_2/CO_2ff} estimates. Our 2022 measurements of the R_{CO_2/CO_2ff} ratios in megacities (Guangzhou and Shenzhen) were consistent with EDGAR estimates (14.9 nmol μmol⁻¹, 2022), while those in smaller cities (Zhanjiang and Shaoguan) were closer to MEIC values (19.2 nmol μmol⁻¹, 2020) (Fig. 6a) and independent field measurements near Xi'an (23 ± 6 nmol μmol⁻¹, 2021) (Liu et al., 2024). City comparisons of observations against MEIC estimates revealed systematic deviations: Shenzhen's observed ratio fell 24.2 % below inventory estimates

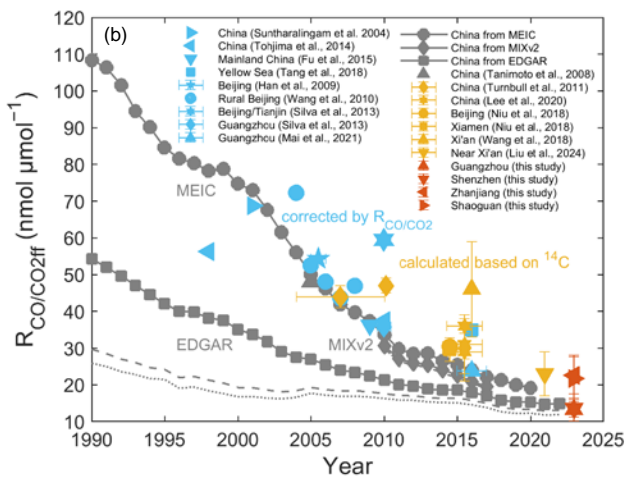
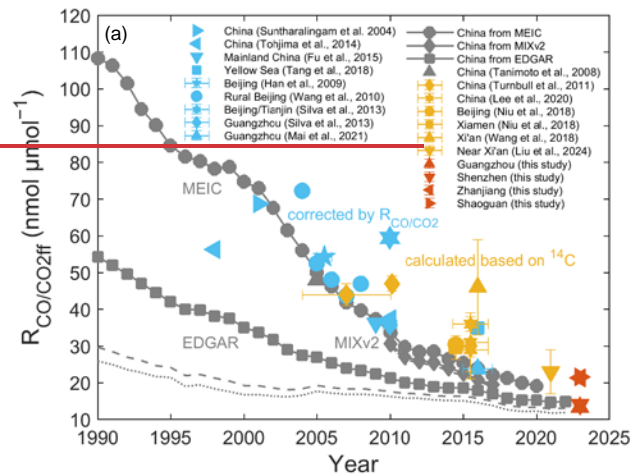
(23.4 nmol μmol^{-1}), whereas Shaoguan's exceeded projections (12.7 nmol μmol^{-1}) by 71.38 %; Guangzhou's and Zhanjiang's are similar to inventory estimates (14.2 and 23.8 nmol μmol^{-1} , respectively) (Fig. 6b).

The three-decade observational $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ ratios are closer to (higher than) the MEIC estimates with a difference of (22 ± 23) % compared with the MIXv2 and EDGAR estimates, when focusing on the ratios over time and ignoring the local deviations caused by the specific cities. These findings indicate that the MEIC inventory is more accurate than the EDGAR inventory for China. For specific cities, we found that the MEIC inventory estimates were deviated less from the observed $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ (based on $\Delta(^{14}\text{C})$ measurements) in recent years than the corrected $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ (using $R_{\text{CO}_2/\text{CO}_2}$) in earlier years for Beijing and Guangzhou (Fig. 6b). For example, in Beijing, the discrepancy in the ratios between observations and inventories decreased from 22 % in 2006–2007 ($R_{\text{CO}_2/\text{CO}_2}$ -corrected) (Wang et al., 2010) to 8.7 % in 2009–2010 ($\Delta(^{14}\text{C})$ -derived) (Turnbull et al., 2011), and further declined to 7.0 % by 2014 ($\Delta(^{14}\text{C})$ -derived) (Niu et al., 2018). Similarly, in Guangzhou, the discrepancy dropped from 84 % in 2009–2010 ($R_{\text{CO}_2/\text{CO}_2}$ -corrected) (Silva et al., 2013) to 34 % in 2014–2017 ($R_{\text{CO}_2/\text{CO}_2}$ -corrected) (Mai et al., 2021), and eventually reached 6.4 % by 2022 ($\Delta(^{14}\text{C})$ -derived). These results suggest that $R_{\text{CO}_2/\text{CO}_2}$ corrections should be carefully interpreted, as the effect of CO_2 from non-fossil sources can significantly bias the results, even in megacities with high C_{ff} emissions. For example, human respiration could bias $R_{\text{CO}_2/\text{CO}_2}$ low by about 9 % at a rural site near Beijing (Wang et al., 2010; Turnbull et al., 2011).

Despite the relatively good agreement of ratios between observations ($R_{\text{CO}_2/\text{CO}_2\text{ff}}$) and MEIC inventory ($I_{\text{CO}_2/\text{CO}_2\text{ff}}$) at the national scale, observational data exhibited significantly greater $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ reduction rates than inventory estimates when examined at the city level. From observations (Fig. 6b), in Guangzhou, $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ decreased by 36 % from 35.8 nmol μmol^{-1} in 2009–2010 (Silva et al., 2013) to 23.8 nmol μmol^{-1} in winter of 2014–2017 (Mai et al., 2021) and by 63 % to 13.3 nmol μmol^{-1} in winter 2022 (partly reflecting seasonal differences, as the Silva et al. (2013) dataset included summer observations, and partly indicating reduced CO emissions relative to C_{ff} due to improved combustion efficiency); in Beijing, $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ decreased by 58 % from 72.3 nmol μmol^{-1} in 2004 (Han et al., 2009) to 30.4 nmol μmol^{-1} in 2014 (Niu et al., 2018); in Xi'an, $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ decreased by 50 % from (46 ± 13) nmol μmol^{-1} in 2016 (Wang et al., 2018) to (23 ± 6) nmol μmol^{-1} in 2021 (Liu et al., 2024). The MEIC estimates for the above three cities decreased by 36 %, 52 %, and 21 %, respectively, over the same period. Larger reductions of the ratios were found from observations than those from the MEIC inventory (i.e., 63 % > 36 % for Guangzhou, 58 % > 52 % for Beijing, and 50 % > 21 % for Xi'an). This conclusion holds even after artificially biasing the $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ ratio downward by about 9 % to account for human respiration in Beijing (2004) and in Guangzhou (2009–2010 and 2014–2017). These findings suggest that the MEIC inventory may insufficiently capture, or lag, the rapid improvement in combustion efficiency and energy structure transformation in China.

The three-decade decline in China's $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ ratios demonstrates both improved fossil fuel combustion efficiency and successful implementation of air pollution control policies i.e., the success of air pollution emission reduction efforts. Our observations reveal significantly greater urban $R_{\text{CO}_2/\text{CO}_2\text{ff}}$ reductions than those estimated by the MEIC inventory, indicating potential underestimation of CO emission reductions relative to C_{ff} mitigations in current inventories. This finding aligns with previous reports of inventory underestimates for real-world CO reductions. Mai et al. (2021) showed that the MEIC

585 inventory may underestimate cumulative reductions from fleet turnover and catalytic converter upgrades, despite China's National V standards having achieved the $\leq 1 \text{ g km}^{-1}$ CO emission limit since 2013. Together, these results imply that the MEIC inventory might systematically underestimate the actual effectiveness of clean air policies in reducing air pollutant emissions.



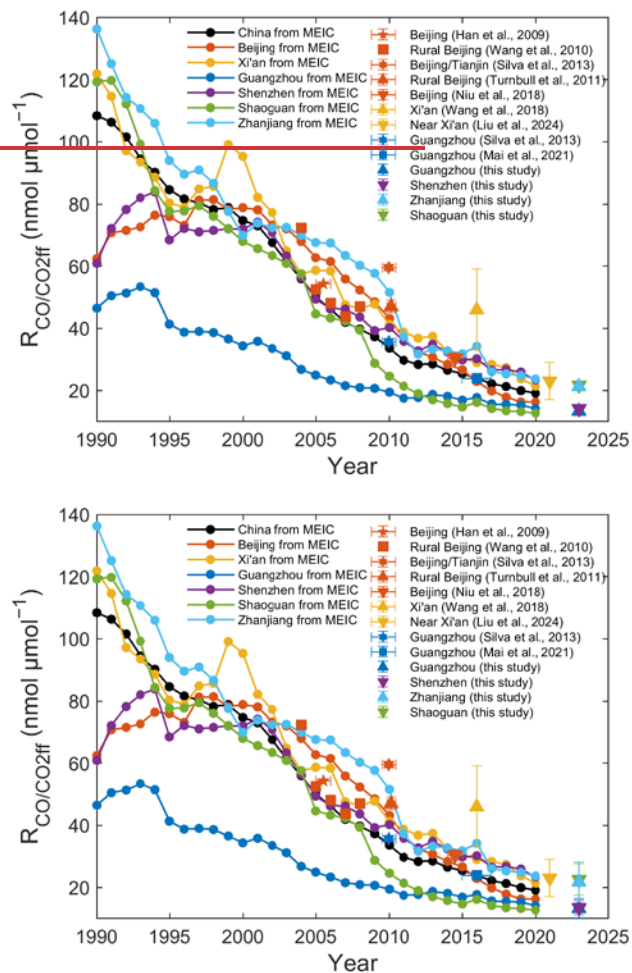


Figure 6: $R_{CO/CO2ff}$ for (a) China and for (b) Chinese cities obtained from inventories and observations (values refer to Table H1). For (a), the gray symbols represent data from the emission inventories (Tanimoto et al., 2008), including MEIC (Meic, 2023; Xu et al., 2024; Li et al., 2019; Li et al., 2017), MIXv2 (Li et al., 2024), and EDGAR2024 (Edgar, 2024). The $I_{CO/CO2ff}$ emission ratios derived from the three inventories are shown with distinct approaches: (1) MEIC calculated the $I_{CO/CO2ff}$ ratio for all

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anthropogenic sectors (represented by solid line with point symbols); (2) MIXv2 computed two variants: combining anthropogenic sectors with open biomass burning (solid line with diamond symbols) and anthropogenic-only emissions (dash-dotted line); while (3) EDGAR2024 provided three ratios: fossil + biogenic CO (solid line with square symbols), fossil + biomass burning CO (dashed line), and fossil-only CO (dotted line), all relative to C_{ff} emissions. The light blue symbols represent $R_{CO/CO2f}$ corrected by $R_{CO/CO2}$ from observational studies (Wang et al., 2010; Tohjima et al., 2014; Suntharalingam et al., 2004; Tang et al., 2018; Han et al., 2009; Fu et al., 2015), assuming that 20 % of the CO_2 enhancement was from sources other than C_{ff} . The orange symbols represent $R_{CO/CO2f}$ calculated based on atmospheric $^{14}CO_2$ measurements from previous studies (Turnbull et al., 2011; Niu et al., 2018; Lee et al., 2020; Wang et al., 2018; Liu et al., 2024). The red symbols depict the values observed in this study. For (b), the Chinese cities include Beijing, Xi'an, Guangzhou, Shenzhen, Zhanjiang, and Shaoguan from the MEIC inventory (filled circles) and observations from previous studies (Wang et al., 2018; Liu et al., 2024; Wang et al., 2010; Silva et al., 2013; Niu et al., 2018; Mai et al., 2021; Han et al., 2009; Turnbull et al., 2011) and this study since 1990. The up and down triangles represent $R_{CO/CO2f}$ estimated based on atmospheric $\Delta(^{14}CO_2)$ measurements. Other symbols represent the $R_{CO/CO2f}$ corrected by $R_{CO/CO2}$ from observational studies, assuming that 20 % of the CO_2 enhancement is from sources other than C_{ff} . The y-axis error bars indicate uncertainty, and the x-axis error bars represent the observed period.

3.6 Implication

Since 2013, China has implemented a series of measures with the explicit aim of improving air quality. While the initial goal of China's clean air targets was to address air pollution, they also served as a powerful catalyst for the simultaneous transformation of energy systems and the mitigation of C_{ff} emissions. As a result, we have observed C_{ff} concentration and emission reductions in some Chinese megacities and supercities, such as Guangzhou, Beijing, and Xi'an. The achievement of peak emissions in Beijing (2010) and Xi'an (2013) marks a pivotal transition for China, signaling that cities across the nation, from megacities to small cities, are gradually reaching their emission peaks. This milestone has profound implications for both China's sustainable development and global climate governance, as China has dominated the global trend since 2010 (Friedlingstein et al., 2023a).

Despite China's remarkable success in reducing C_{ff} emissions, continued efforts are needed to optimize the nation's energy system and economic structure in order to facilitate future green growth. It is imperative that common solutions to climate change and air pollution are formulated and implemented with urgency, as China has set a goal for all cities to meet current air quality standards by 2035 and has pledged to achieve carbon peak by 2030 and carbon neutrality by 2060. One available solution is to control the common key sources and dominant source regions of air pollution and CO_2 emissions (Wu et al., 2022; Zheng et al., 2024). In future policymaking, it is essential to adopt a co-beneficiary strategy that co-ordinates clean air measures and addresses climate change measures. This strategy, together with the associated assessment approach, will be an essential part of achieving sustainable development.

4 Conclusions and outlook

This study advances the understanding of urban C_{ff} concentration changes in China through three key contributions. First, we pioneer the integration of biomass burning emissions into C_{ff} estimation frameworks, significantly improving methodological accuracy. Second, we identify substantial C_{ff} reductions in cities and their source regions, driven by coal-to-

gas transitions (evidenced by stable isotope analysis) and combustion efficiency improvements (confirmed by declining R_{CO_2/CO_2ff} ratios), where megacities and supercities lead this decline. Finally, through systematic analysis of long-term R_{CO_2/CO_2ff} trends, we reveal current emission inventories may underestimate combustion efficiency gains and CO emission reductions relative to C_{ff} mitigations. These findings provide critical support for refining emission accounting systems and developing evidence-based climate policies. The integrated approach offers new insights into urban emission dynamics and mitigation effectiveness.

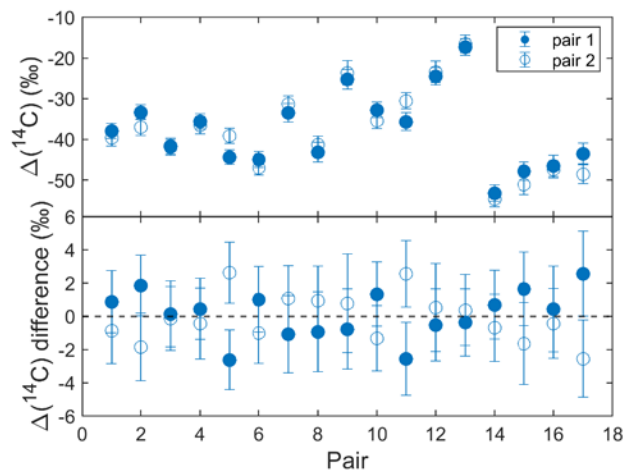
This study has some limitations in sampling and source attribution. First, current sampling only covers summer and winter; future work should include all seasons to better capture annual trends. Second, ~~the $\delta(^{13}C)$ -based source partitioning is associated with large uncertainties—on the order of tens of percent—due to the limited isotopic separation among CO_2 sources and the poorly constrained biogenic endmember. Similar uncertainty ranges have been reported in previous urban studies (see Table II). Therefore, the $\delta(^{13}C)$ partitioning results presented here should be considered as a preliminary, first-order estimate. Signatures used here mainly rely on published data, introducing potential uncertainties.~~ Direct measurements of source-specific isotopic values would help refine the analysis.

In future work, a detailed quantitative analysis linking C_{ff} to emission distributions using FLEXPART footprints will be conducted to provide a more rigorous connection between observations and emission sources. Additionally, future studies should explicitly consider seasonally varying background references, ideally including coastal or marine background sites to better represent summer air masses. Furthermore, upcoming efforts could incorporate atmospheric modelling and inversion methods to improve emission estimates. This would require high resolution prior flux data and validation against direct measurements (e.g., radiocarbon analysis). Addressing these gaps would enhance source apportionment accuracy and enable a more robust integration of top-down (e.g., inversions) and bottom-up (e.g., inventories) approaches for evaluating urban emission mitigation strategies.

Appendix A: Seasonal averages and quality control of $\Delta(^{14}\text{C})$ and $\delta(^{13}\text{C})$ measurements

Table A1 $\Delta(^{14}\text{C})$ and $\delta(^{13}\text{C})$ averages and standard deviations ($n=4$ for each value) at 30 sampling sites

| City | Site code | Summer | | Winter | | Altitude (m a.s.l.) | Elevation (m a.g.l.) | Site description |
|-----------|------------|-----------------------------|-----------------------------|-----------------------------|-----------------------------|---------------------|----------------------|--|
| | | $\Delta(^{14}\text{C})$ (‰) | $\delta(^{13}\text{C})$ (‰) | $\Delta(^{14}\text{C})$ (‰) | $\delta(^{13}\text{C})$ (‰) | | | |
| Guangzhou | GZ1 | -23.8 ± 9.8 | -9.0 ± 0.7 | -46.0 ± 8.1 | -8.9 ± 0.2 | 212 | 25 | Suburban rooftops |
| | GZ2 | -30.2 ± 4.7 | -9.0 ± 0.3 | -56.8 ± 5.6 | -8.9 ± 0.3 | 19 | 20 | Suburban rooftops |
| | GZ3 | -24.9 ± 8.8 | -8.7 ± 0.3 | -46.5 ± 3.9 | -8.7 ± 0.2 | 120 | 30 | Suburban rooftops |
| | GZ4 | -24.6 ± 4.8 | -9.0 ± 0.7 | -49.7 ± 9.3 | -9.0 ± 0.3 | 23 | 35 | Urban rooftops |
| | GZ5 | -42.6 ± 12.2 | -9.6 ± 0.8 | -48.0 ± 8.3 | -8.5 ± 0.3 | 46 | 35 | Urban rooftops |
| | GZ6 | -42.1 ± 16.7 | -9.5 ± 1.0 | -52.8 ± 11.2 | -8.9 ± 0.1 | 53 | 60 | Urban rooftops |
| | GZ7 | -25.5 ± 5.0 | -8.8 ± 0.4 | -48.1 ± 7.3 | -8.9 ± 0.3 | 120/ 75 | 118/ 40 | Urban tower/ Urban rooftops |
| | GZ8 | -32.8 ± 9.4 | -9.4 ± 0.5 | -49.6 ± 8.1 | -9.2 ± 0.2 | 12 | 30 | Urban rooftops |
| | GZ9 | -35.8 ± 5.4 | -9.8 ± 0.4 | -51.1 ± 6.9 | -8.7 ± 0.3 | 50 | 30 | Urban rooftops |
| | GZ10 | -34.8 ± 19.8 | -9.5 ± 0.9 | -51.2 ± 5.9 | -9.0 ± 0.2 | 54 | 40 | Suburban rooftops |
| Shenzhen | SZ1 | -23.4 ± 18.0 | -8.6 ± 0.5 | -46.3 ± 27.0 | -9.2 ± 0.2 | 40 | 30 | Suburban tower |
| | SZ2 | -63.4 ± 3.9 | -9.1 ± 0.4 | -51.9 ± 8.9 | -9.4 ± 0.1 | 28 | 15 | Rooftops in Industrial area |
| | SZ3 | -20.8 ± 22.5 | -8.7 ± 0.5 | -56.3 ± 16.1 | -9.4 ± 0.3 | 14 | 15 | Urban rooftops |
| | SZ4 | -23.7 ± 27.8 | -8.8 ± 0.6 | -53.7 ± 15.2 | -9.5 ± 0.3 | 42 | 40 | Urban campus rooftops |
| | SZ5 | -26.4 ± 17.5 | -8.8 ± 0.3 | -50.5 ± 12.4 | -9.4 ± 0.3 | 40 | 30 | Urban campus rooftops |
| | SZ6 | -25.8 ± 10.3 | -8.7 ± 0.5 | -51.4 ± 31.9 | -9.1 ± 0.4 | 60 | 30 | Suburban tower |
| | SZ7 | -19.8 ± 6.1 | -8.8 ± 0.1 | -48.6 ± 33.2 | -9.3 ± 0.6 | 210 | 200 | Urban rooftops |
| | SZ8 | -18.4 ± 12.7 | -8.6 ± 0.5 | -43.8 ± 31.7 | -9.0 ± 0.3 | 150 | 110 | Suburban rooftops at the boundary site |
| | SZ9 | -13.1 ± 11.4 | -8.5 ± 0.4 | -45.1 ± 32.0 | -9.2 ± 0.3 | 60 | 30 | Suburban tower |
| | SZ10 | 6.8 ± 1.9 | -9.0 ± 0.6 | -52.8 ± 7.7 | -9.4 ± 0.2 | 60 | 20 | Suburban rooftops |
| Zhanjiang | ZJ1 | -19.1 ± 9.4 | -10.2 ± 0.5 | -35.7 ± 12.9 | -9.1 ± 0.4 | 8 | 20 | Rural rooftops |
| | ZJ2 | -23.5 ± 14.6 | -9.4 ± 0.6 | -35.5 ± 12.6 | -9.0 ± 0.2 | 24 | 40 | Urban rooftops |
| | ZJ3 | -18.4 ± 4.8 | -10.2 ± 0.6 | -37.0 ± 12.7 | -9.1 ± 0.1 | 44 | 40 | Urban rooftops |
| | ZJ4 | -14.2 ± 7.0 | -8.9 ± 0.3 | -38.7 ± 12.3 | -9.1 ± 0.2 | 25 | 40 | Urban rooftops |
| | ZJ5 | -18.0 ± 13.1 | -9.1 ± 0.5 | -40.3 ± 11.7 | -9.4 ± 0.4 | 41/ 46 | 50/ 30 | Suburban campus site/ Site near the port area |
| Shaoguan | SG1 | -28.3 ± 15.1 | -8.9 ± 0.3 | -20.3 ± 5.1 | -9.0 ± 0.1 | 114 | 30 | Suburban rooftops |
| | SG2 | -29.2 ± 15.0 | -9.5 ± 0.3 | -28.2 ± 3.8 | -8.9 ± 0.2 | 60 | 40 | Urban campus rooftops |
| | SG3 | -26.5 ± 17.3 | -9.2 ± 0.5 | -35.0 ± 10.3 | -9.0 ± 0.2 | 68 | 40 | Urban rooftops |
| | SG4 | -43.6 ± 20.8 | -9.0 ± 0.2 | -19.7 ± 6.2 | -8.8 ± 0.2 | 95 | 30 | Rural site |
| | SG5/ NL | -3.75 ± 1.3 | -9.2 ± 0.2 | -10.92 ± 0.8 | -9.3 ± 0.2 | 1700 | 15 | Rooftops at the background site |



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Figure A1 Pair differences of $\Delta(^{14}\text{C})$ for replicate measurements. Replicates were obtained from parallel air samples. The difference of each individual measurement from its pair mean is shown. Closed and open symbols are the first and second group taken from each pair, respectively. Error bars are the 1-sigma uncertainty on each measurement.

665 Appendix B: Radiocarbon isotope endmembers for biomass burning

Atmospheric $^{14}\text{CO}_2$ is assimilated by plants via photosynthesis, imprinting atmospheric $\Delta(^{14}\text{CO}_2)$ signatures into plant tissues. This creates a bidirectional link: plant $\Delta(^{14}\text{C})$ reflects atmospheric $\Delta(^{14}\text{CO}_2)$ levels, while atmospheric $\Delta(^{14}\text{CO}_2)$ dynamics can be inferred from plant biomass archives (e.g., tree rings). Annual biomass $\Delta(^{14}\text{C})$ closely matches contemporaneous atmospheric $\Delta(^{14}\text{CO}_2)$ (due to rapid carbon turnover within a single growing season). Multi-year biomass $\Delta(^{14}\text{C})$ represents an integrated signal, blending atmospheric $\Delta(^{14}\text{CO}_2)$ variations over its growth period (e.g., tree rings capture annual $\Delta(^{14}\text{CO}_2)$ fluctuations).

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B1 Annual biomass. The $\Delta(^{14}\text{C})$ for annual biomass (Δ_a) in 2022 was estimated as -14.8 ± 2.2 ‰ (mean \pm MSE \pm Δ_a), derived from a linear regression model of atmospheric $\Delta(^{14}\text{CO}_2)$ decline (-4.4 ‰ a^{-1}) observed in Northern Hemisphere zone 3 between 2010 and 2018 (Hua et al., 2021).

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B2 Multi-year biomass. The $\Delta(^{14}\text{C})$ for multi-year biomass (Δ_m) is related with its age; the year it was growing, the annual increase in biomass, and atmospheric $^{14}\text{CO}_2$ during its growth cycle. The $\Delta(^{14}\text{C})$ for multi-year biomass can be determined (Lewis et al., 2004):

$$\Delta^{14}C_m C = \frac{\int_{t_1}^{t_2} \Delta^{14}C(t) w(t) dt}{\int_{t_1}^{t_2} w(t) dt} \quad (B1)$$

where $\Delta^{14}C(t)$ is the atmospheric $\Delta^{14}C(t)$ at age t , and the weighting function $w(t)$ is the growth rate of carbon in biomass at age t , which can be determined by the Chapman-Richards growth model (Lewis et al., 2004):

$$V = A(1 - e^{-\frac{t-t_0}{\tau}})^m \quad (B2)$$

$$w(t) = \frac{dV}{dt} \quad (B3)$$

where V is the volume of a tree at age t ($V = 0$ at $t = t_0$), and the parameters A , τ , and m can be chosen empirically to fit measured tree growth characteristics. The Chapman-Richards growth model describes cumulative growth of V .

It is assumed that the multi-year biomass was partitioned into five age cohorts (10-, 20-, 40-, 65-, and 85-year-old trees) with relative share of $20 \pm 10\%$, $20 \pm 10\%$, $40 \pm 20\%$, $10 \pm 5\%$ and $10 \pm 5\%$, respectively (Mohn et al., 2008). The corresponding $\Delta^{14}C$ values were calculated as $20.9 \pm 5.4\%$, $52.9 \pm 4.0\%$, $137.5 \pm 35.1\%$, $261.2 \pm 50.4\%$, and $203.1 \pm 17.4\%$, respectively. Consequently, the $\Delta^{14}C$ signature of the multi-year biomass for the year 2022 was estimated as $116.2 \pm 17.6\%$ (mean $\pm 1\sigma$; Δ_m) using the Chapman-Richards growth model ($\tau = 50$, $m = 3$) and long-term tree-ring $\Delta^{14}C$ measurements (Hua et al., 2021).

B3 Biomass burning. The $\Delta^{14}C$ endmember for biomass burning (Δ_{BB}) was calculated using the two biomass types:

$$\Delta_{BB} = f_a \cdot \Delta_a + (1 - f_a) \cdot \Delta_m \quad (B4)$$

where Δ_a and Δ_m represent the $\Delta^{14}C$ signatures of annual biomass (e.g., crop residues) and multi-year biomass (e.g., woody waste), respectively, and f_a is the annual biomass fraction.

Using this framework, we estimated the 2022 $\Delta^{14}C$ endmembers for biomass burning as $116.2 \pm 17.6\%$, $103.1 \pm 15.8\%$, $90.0 \pm 14.1\%$, $76.9 \pm 12.3\%$, $63.8 \pm 10.6\%$, and $50.7 \pm 8.9\%$ for f_a values of 0%, 10%, 20%, 30%, 40%, 50%, respectively.

Appendix C: Bias correction for C_R calculation

C1 Air-sea exchange. The potential influence of CO_2 outgassing from the adjacent South China Sea (SCS) on our onshore measurements was assessed. Although the SCS is a net source of CO_2 to the atmosphere (with an annual flux of $0.44 \text{ mol m}^{-2} \text{ yr}^{-1}$ (Li et al., 2020)), its influence is negligible. This conclusion is supported by an analogous study of the California coast: the high-resolution WRF-STILT simulation by Graven et al. (2018) was conducted using flux data that included intense local nearshore sources (with fluxes up to $1.11 \text{ mol m}^{-2} \text{ yr}^{-1}$ (Turi et al., 2014)). Their results demonstrated that even these potent sources altered onshore CO_2 concentrations by less than 0.001 ppm (Graven et al., 2018). Given that the regional net flux from the SCS is weaker than this analogue, we conclude its impact on our $\Delta^{14}CO_2$ measurements and derived C_R estimates is physically insignificant and within the measurement uncertainty.

C2 Nuclear facilities. All operational (Daya Bay, Ling'ao, Yangjiang, Taishan) and under-construction (Lufeng, Taipingling, Lianjiang) nuclear power plants (NPPs) along the ~~eastline of~~ Guangdong Province ~~coastline~~ (Table C1) employ pressurized water reactor (PWR) technology. Airborne ^{14}C releases from these facilities are predominantly hydrocarbons (75–95 %), mainly CH_4 , with only a small fraction ~~emitted as in the form of~~ CO_2 (Iaea, 2004). In fact, almost all ~~commercial~~ reactors in China (> 95 %) are ~~of~~ the PWR type, which ~~exhibits~~ the lowest $^{14}\text{CO}_2$ emission factor ~~for~~ $^{14}\text{CO}_2$ ~~release among nuclear technologies~~ (Graven and Gruber, 2011; Yang, 2024).

Graven and Gruber (2011) ~~not reported~~ that most of China and the western US are ~~region~~ the areas with ~~minimal~~ very little potential bias in ~~the derived~~ $^{14}\text{CO}_2$ -based C_{ff} estimates, owing to ~~strong~~ intense fossil fuel ~~signals~~ emissions and ~~limited~~ but little to no nuclear ^{14}C influence activity. Consistently, Graven et al. (2018) simulated the impact of reactor emissions on atmospheric $\Delta^{14}\text{CO}_2$ using WRF-STILT and found that an average $^{14}\text{CO}_2$ release rate of 6.6 Ci yr⁻¹ (~0.24 TBq yr⁻¹) from the Diablo Canyon NPP in California produced an effect of < 0.1 ppm in inferred C_{ff} at all sites, confirming this view that nuclear ^{14}C emissions from reactors in the western US have a negligible influence effect on atmospheric radiocarbon measurements their samples, ~~simulating an effect on inferred C_{ff} of less than 0.1 ppm.~~

In Guangdong Province, Zazzeri et al. (2018) estimated $^{14}\text{CO}_2$ emissions from the Daya Bay, Ling'ao, and Yangjiang NPPs to be 0.111, 0.233, and 0.166 TBq yr⁻¹, respectively, values comparable to or smaller than those reported for Diablo Canyon. Although Daya Bay and Ling'ao are located only 6–7 km from the nearest observation sites (SZ10 and SZ9) (Table C1), their emission rates remain extremely low. Under prevailing southeasterly winds in summer and northeasterly winds in winter, dispersion within the coastal boundary layer further dilutes any potential $^{14}\text{CO}_2$ plumes before they reach the sampling locations.

Based on Gaussian plume scaling and regional wind climatology, we estimate that the contribution of local reactor $^{14}\text{CO}_2$ to measured $\Delta^{14}\text{CO}_2$ at these urban sites would be < 0.1 ‰, corresponding to an effect on inferred C_{ff} below 0.05 ppm. Therefore, even for the closest stations, the impact of nearby nuclear facilities on $\Delta^{14}\text{CO}_2$ measurements is considered negligible and does not affect our radiocarbon-based source partitioning. ~~Although we can't obtain the $A(^{14}\text{CO}_2)$ emission monitoring data from NPPs in China, National Report on Radiation Environment Quality confirms that no incidents of excessive radiation dose discharges have occurred at NPPs in Guangdong Province. For example, data from 2011 to 2014 show that liquid radionuclide discharges from Daya Bay and Ling'ao NPPs were only 0.09–0.38 % of legal thresholds. The 2019 report highlighted that public radiation doses at NPPs such as Daya Bay and Taishan remained "significantly below regulatory limits" (Meece, 2021). We thus assume that the $^{14}\text{CO}_2$ effect from the NPPs in Guangdong Province may also be negligible, referred to the results from the western US and Chinese regulatory standards for radioactive discharges from NPPs.~~

Table C1 Summary of operational nuclear facility in Guangdong Province: installed capacity, operational period, $^{14}\text{CO}_2$ emissions, and proximity to sampling sites with dominant wind directions.

| Nuclear facility | Installed electric capacity ^a (MWe) | Operation period ^b | $^{14}\text{CO}_2$ emissions in 2016 ^c (TBq yr ⁻¹) | Closest site | Distance (km) | Dominant wind direction |
|------------------|--|-------------------------------|---|--------------|---------------|-------------------------|
|------------------|--|-------------------------------|---|--------------|---------------|-------------------------|

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|-------------------------------|--|---|------------------------------------|--|---------------------------------------|--|
| Daya Bay 1-2 | 984/984 | 1994/1994-present | 0.111 | SZ10/ SZ9 | 6/ 22 | SE (Summer) NE (Winter) |
| Ling'ao 1-4 | 990/990/ 1086/1086 | 2022/2003/ 2010/2011-present | 0.233 | SZ10/ SZ9 | 7/ 22 | |
| Yangjiang 1-6 | 1086/1086/1086/ 1086/1086/1086 | 2014/2015/2016/ 2017/2018/2019- present | 0.166/ unknown | ZJ1/ GZ10/ SZ3 | 164/ 182/ 191 | |
| Taishan 1-2 | 1750/1750 | 2018/2019-present | unknown | GZ10/ SZ3/ ZJ1 | 114/ 116/ 242 | |

740 ^a Data from China Nuclear Energy Association, *Operational Performance of Nuclear Power in China (January-December 2024)* (in Chinese). Retrieved from: <https://www.china-nea.cn/site/content/48480.html> (last access: 18 October 2025). ^b Data from National Nuclear Safety Administration, *The Status of Mainland China's Nuclear Power Units in 2024* (in Chinese). Retrieved from: https://nnsa.mee.gov.cn/ywdt/hyzz/202501/t20250107_1100142.html (last access: 18 October 2025). ^c Emission data from [Zazzeri et al. \(2018\)](#), estimated using emission factors multiplied by IAEA PRIS data, assuming 28 % of

745 ¹⁴C released from PWRs is in the form of CO₂.

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C32 Biospheric exchange. Biospheric carbon fluxes associated with photosynthesis, autotrophic respiration, and annual biomass burning generally do not alter atmospheric $\Delta(^{14}\text{C})$ levels, as the carbon exchanged through these processes largely maintains isotopic equilibrium with contemporary atmospheric CO₂ (Turnbull et al., 2009). In contrast, heterotrophic respiration and multi-year biomass burning (e.g., wildfire consuming legacy organic matter) release carbon fixed during periods of elevated atmospheric $\Delta(^{14}\text{C})$, such as the 1960s nuclear bomb testing peak. This temporal decoupling between carbon uptake and release introduces a measurable positive bias in modern $\Delta(^{14}\text{C})$, reflecting the delayed contribution of older carbon pools. Therefore, we use estimates of heterotrophic respiration (Rh) and biomass burning (BB) fluxes to correct for biospheric influence on C_{ff} calculation.

755 **C32.1 Heterotrophic respiration.** The heterotrophic respiration correction term (β_{Rh}) is calculated by the following equation:

$$\beta_{\text{Rh}} = \frac{C_{\text{Rh}}(\Delta_{\text{bg}} - \Delta_{\text{Rh}})}{\Delta_{\text{bg}} + 1000\text{‰}} \quad (\text{C1})$$

760 where C_{Rh} is the CO₂ mole fraction estimated by coupling hourly FLEXPART footprints with the heterotrophic respiration fluxes extracted from the Carnegie Ames Stanford Approach Global Fire Emissions Database Version 4 (CASA-GFED4s) (Randerson et al., 2017; Van Der Werf et al., 2017). We imposed the diurnal cycle from the CASA-GFED3 (Van Der Werf et al., 2010) heterotrophic respiration fluxes (estimated as half of the ecosystem respiration, which is calculated as the difference between net ecosystem exchange and gross ecosystem exchange; [NEE - GEE]/2) onto the nearest neighbor CASA-GFED4s monthly mean fluxes to approximate hourly resolved fluxes. By aggregating these flux estimates, we created flux maps matching the spatial resolution of the hourly FLEXPART footprints. We then calculated C_{Rh} by

765 multiplying the FLEXPART footprints with heterotrophic respiration flux maps. The simulated C_{Rh} concentrations were $1.5 \pm 0.7 \mu\text{mol mol}^{-1}$ (range: $0.5\text{--}3.1 \mu\text{mol mol}^{-1}$) in summer and $2.2 \pm 0.9 \mu\text{mol mol}^{-1}$ (range: $0.4\text{--}3.8 \mu\text{mol mol}^{-1}$) in winter. We used a value of $40 \pm 35 \%$ for the $\Delta(^{14}\text{C})$ signature of heterotrophic respiration (Δ_{Rh}), based on the value of $75 \pm 35 \%$ in 2015 (Graven et al., 2018) and considering a decrease of 5 % per year (Zazzeri et al., 2023). The disequilibrium correction from heterotrophic respiration (β_{Rh}) were estimated to be $-0.06 \pm 0.03 \text{ ppm}$ (range: -0.14 to -0.02 ppm) in summer and $-0.11 \pm 0.04 \text{ ppm}$ (range: -0.20 to -0.02 ppm) in winter.

770 **C3.2.2 Biomass burning.** For the influence of biomass burning, we compared extracted CO_2 emissions from two datasets: biomass burning in Guangdong Province and four cities from the CASA-GFED4s (Randerson et al., 2017; Van Der Werf et al., 2017), and $C_{\text{bio_edgar}}$ from the Emissions Database for Global Atmospheric Research (EDGAR) (Edgar, 2024). The key methodological distinctions lies in their differing scopes of these datasets: CASA-GFED4s quantifies emissions from open-environment fires that are detectable by (i.e., satellite-observable combustion events including wildfires, agricultural residue burning, savanna/rangeland fires, and other small-scale open burning events, fires). In contrast, whereas EDGAR $C_{\text{bio_edgar}}$ also represents anthropogenic biofuel combustion, such as incorporates emissions from closed-system combustion (e.g., industrial and/or residential biomass use), while explicitly excluding large-scale wildfires and land-use change-related emissions (LULUCF). Thus, the two datasets characterize different aspects of biomass combustion: CASA-GFED4s captures open burning, whereas EDGAR focuses on controlled, human-induced combustion. Across Guangdong Province and the four studied cities, biomass burning emissions from CASA-GFED4s accounted for <2 % of R_h emissions, indicating a minor contribution from open-environment fires in this region across Guangdong Province and the four studied cities. By contrast, EDGARv2024ghg $C_{\text{bio_edgar}}$ estimates represented a much substantially larger proportion of R_h emissions, ranging from 7–29 % (Guangdong), 24–92 % (Guangzhou), 16–97 % (Shenzhen), 13–38 % (Zhanjiang), and to 73–248 % (Shaoguan). Given the small disequilibrium corrections for R_h , the simulated contribution of BB emissions to C_{f} estimates was negligible magnitude of when using CASA-GFED4s emissions and their minimal effect on C_{f} quantification, we focused on the EDGAR $C_{\text{bio_edgar}}$ dataset for subsequent analysis. This dataset provides a more complete and representative estimate of anthropogenic biomass combustion CO_2 emissions in urban areas, which are most relevant to our study objectives. However, EDGAR-derived C_{bio} scenarios indicated a potentially higher influence on C_{f} quantification, particularly in regions with high biomass combustion activity.

790 To estimate the biomass burning correction term (β_{BB}) using the EDGAR2024 C_{bio} inventory (Edgar, 2024), we first derived total C_{BB} simulations (C_{BB}) by applying a biomass burning fraction (α_{BB}) to the $C_{\text{bio_edgar}}$ simulations ($C_{\text{bio_edgar}}$). This parameter α_{BB} represents the proportion of $C_{\text{bio_edgar}}$ emissions attributable to biomass burning:

$$C_{\text{BB}} = C_{\text{bio_edgar}} \cdot \alpha_{\text{BB}} \quad (\text{C2})$$

795 The correction term β_{BB} was subsequently calculated as:

$$\beta_{\text{BB}} = \frac{C_{\text{BB}}(\Delta_{\text{bg}} - \Delta_{\text{BB}})}{\Delta_{\text{bg}} + 1000\%} \quad (\text{C3})$$

For simulated $C_{\text{bio_edgar}}$ mole fraction estimation in 2022, we implemented a three-stage process: (1) Generating $0.1^\circ \times 0.1^\circ$ resolution flux maps through integration of EDGAR2024 $C_{\text{bio_edgar}}$ emission fluxes with FLEXPART atmospheric transport footprints, (2) performing spatiotemporal aggregation to align with FLEXPART model output specifications, and (3) calculating concentrations via convolution operations between transport footprints and optimized flux fields.

We adopted $\Delta(^{14}\text{CO}_2)$ signatures of $-14.8 \pm 2.2 \text{ ‰}$ (annual biomass burning) and $116.2 \pm 17.6 \text{ ‰}$ (multi-year biomass burning) calculated in Appendix B. For 2022 $\Delta(^{14}\text{C})$ endmembers for biomass burning, we estimated values of $116.2 \pm 17.6 \text{ ‰}$ (0 % annual biomass), $103.1 \pm 15.8 \text{ ‰}$ (10 %), $90.0 \pm 14.1 \text{ ‰}$ (20 %), $76.9 \pm 12.3 \text{ ‰}$ (30 %), $63.8 \pm 10.6 \text{ ‰}$ (40 %), and $50.7 \pm 8.9 \text{ ‰}$ (50 %), corresponding to incremental annual biomass burning fractions from 0 % to 50 %.

We quantified disequilibrium correction terms under the maximum biomass burning (BB) contribution scenario ($f_a = 0 \text{ ‰}$ and $\alpha_{\text{BB}} = 100 \text{ ‰}$). The simulated C_{BB} concentrations were $0.8 \pm 0.7 \text{ } \mu\text{mol mol}^{-1}$ (range: $0.0\text{--}1.9 \text{ } \mu\text{mol mol}^{-1}$) in summer and $1.9 \pm 1.0 \text{ } \mu\text{mol mol}^{-1}$ (range: $0.1\text{--}4.4 \text{ } \mu\text{mol mol}^{-1}$) in winter. The BB-specific correction term (β_{BB}) exhibited seasonal variations: $-0.09 \pm 0.08 \text{ } \mu\text{mol mol}^{-1}$ (range: -0.46 to $-0.01 \text{ } \mu\text{mol mol}^{-1}$) in summer and $-0.24 \pm 0.12 \text{ } \mu\text{mol mol}^{-1}$ (range: -0.56 to $-0.02 \text{ } \mu\text{mol mol}^{-1}$) in winter under 0 % annual biomass burning contribution. The combined correction factor β , integrating contributions from both heterotrophic respiration (Rh) and biomass burning (BB), showed broader ranges: $-0.16 \pm 0.09 \text{ } \mu\text{mol mol}^{-1}$ (range: -0.55 to $-0.03 \text{ } \mu\text{mol mol}^{-1}$) in summer and $-0.35 \pm 0.15 \text{ } \mu\text{mol mol}^{-1}$ (range: -0.72 to $-0.04 \text{ } \mu\text{mol mol}^{-1}$) in winter.

Appendix D: Background selection

As we summarized in Zhou et al. (2024), Turnbull et al. (2015) concluded that for Indianapolis, a city with relatively simple boundary conditions, the upwind background site (Tower 1) is more appropriate compared with continental and regional background sites (LEF and NWR). In contrast, for Los Angeles, a city with relatively complex boundary conditions, Newman et al. (2016) and Miller et al. (2020) tend to use the neighboring regional or continental background sites (MWO and LJO; BRW and NWR), because the upwind background within the city may be influenced by emissions from neighboring cities and therefore cannot represent the local urban background.

In this study, the cities concerned are central cities or neighboring cities of the Pearl River Delta (PRD) urban agglomeration with relatively complex boundary conditions, so we chose the nearest regional background site NL (i.e., SG5) as the background for determining the C_{ff} concentrations.

The Nanling site serves as an ideal site for Guangdong's atmospheric background monitoring station for Guangdong due to its remote location ($>100 \text{ km}$ from the PRD urban agglomeration), high-altitude terrain ($>1700 \text{ m a.s.l.}$) avoiding localized pollution. Situated above the mixed boundary layer under most meteorological conditions, NL is well isolated from urban influences. Its strategic position as a climatic and watershed boundary intercepting seasonal airflows further collectively enabling precise monitoring of regional atmospheric transport patterns while meeting strict background-station criteria for pollution isolation and cross-boundary impact assessment.

830 In addition, trajectory and potential source region analyses further confirm that the Nanling site is representative of regional background air under both monsoon regimes. HYSPLIT clustering and PSCF analyses by Zhang et al. (2022) showed that during winter, air masses arriving at Nanling predominantly originate from the northern inland provinces (e.g., Hunan, Jiangxi, Sichuan), while in summer they mostly come from the South China Sea and southeastern coastal regions. These summer air masses are typically marine-influenced and low in CO₂, consistent with clean background characteristics. Continuous CO observations at NL further support its background representativeness: approximately 90 % of summer samples had concentrations below 200 ppb, comparable to other regional background sites, indicating that pollution from the PRD rarely reaches NL. Therefore, NL can be considered a robust regional background site for both monsoon seasons.

835 Furthermore

840 On the other hand, the “annual” CO₂ and $\Delta(^{14}\text{C})$ averages at NL, which are close to those at the Jungfraujoch, occupy a position in the upper-right section of the Keeling plot of $\Delta(^{14}\text{C})$ versus CO₂, representing the background end-member when, by comparison with the values at the Waliguan (Fig. 3). Additionally, the $\Delta(^{14}\text{C})$ and CO₂ averages at NL were the highest and the lowest, respectively, among the 30 sampling sites, respectively (Table A1), which are consistent with background-level characteristics.

Appendix E: Comparison of C_{ff} fractions and concentrations among various cities

845 **Table E1** Comparison of C_{ff} and C_{bio} fractions derived from $\Delta(^{14}\text{CO}_2)$ measurements in various cities

| City | Time | Background | C_{ff} (%) | C_{bio} (%) | References |
|-----------------|------------------|------------------|--------------|---------------|-----------------------|
| Paris | 2010 | MHD ^a | 77 | 23 | (Lopez et al., 2013) |
| Los Angeles | 2006-2013 winter | LJO ^b | 86 | 14 | (Newman et al., 2016) |
| Los Angeles | 2006-2013 summer | LJO | 93 | 7 | (Newman et al., 2016) |
| Los Angeles | 2014.11-2016.03 | MWO ^c | 80 | 20 | (Miller et al., 2020) |
| Beijing | 2014 | WLG ^d | 75.2 ± 14.6 | 24.8 | (Niu et al., 2016) |
| Xiamen | 2014 | WLG | 59.1 ± 26.8 | 40.9 | (Niu et al., 2016) |
| Xi'an | 2014 winter | WLG | 92.7 ± 9.7 | 7.3 ± 9.7 | (Zhou et al., 2020) |
| Xi'an, urban | 2016 summer | WLG | 82.5 ± 23.8 | 17.5 | (Wang et al., 2018) |
| Xi'an, urban | 2016 winter | WLG | 61.8 ± 10.6 | 38.2 | (Wang et al., 2018) |
| Xi'an, suburban | 2016 summer | WLG | 90.0 ± 24.8 | 10.0 | (Wang et al., 2018) |
| Xi'an, suburban | 2016 winter | WLG | 57.4 ± 9.7 | 42.6 | (Wang et al., 2018) |
| Guangzhou | 2022 winter | NL ^e | 79 ± 5 | 21 | this study |
| Shenzhen | 2022 winter | NL | 73 ± 6 | 27 | this study |
| Zhanjiang | 2022 winter | NL | 59 ± 4 | 41 | this study |
| Shaoguan | 2022 winter | NL | 53 ± 13 | 47 | this study |

^a Mace Head, ^b La Jolla, ^c Mount Wilson Observatory, ^d Waliguan, ^e Nanling

850 **Table E2** Comparison of C_{ff} concentrations derived from $\Delta(^{14}\text{CO}_2)$ measurements in various cities

| Country | City | Sampling period [†] Time | Sampling time | Duration | C_{ff} ($\mu\text{mol mol}^{-1}$) | References |
|----------------|-----------------|-----------------------------------|--------------------|----------------|---------------------------------------|-----------------------------|
| Poland | Krakow | 1989 | == | <u>2 weeks</u> | 27.5 | (Kuc et al., 2003) |
| Poland | Krakow | 1994 | == | <u>2 weeks</u> | 10 | (Kuc et al., 2003) |
| Poland | Krakow | 2005–2009 | == | <u>1 week</u> | 1.98–2.18 | (Zimnoch et al., 2012) |
| Poland | Kasprowy Wierch | 2005–2009 | == | <u>1 week</u> | 1.95–2.08 | (Zimnoch et al., 2012) |
| Poland | Gliwice | 2011.01–2013.01 | <u>10:00</u> | == | 23–24 | (Piotrowska et al., 2020) |
| Czech Republic | Prague | 2001–2018 | == | <u>1 month</u> | 25.51 \pm 11.45 | (Svetlik et al., 2010) |
| Slovakia | Bratislava | 1999–2007 | == | <u>1 month</u> | 25.56 \pm 6.90 | (Svetlik et al., 2010) |
| Germany | Heidelberg | 1986–1996 | == | <u>2 weeks</u> | 11.09 \pm 0.24 | (Levin and Rödenbeck, 2008) |
| Germany | Heidelberg | 1997–2007 | == | <u>2 weeks</u> | 10.92 \pm 0.34 | (Levin and Rödenbeck, 2008) |
| Hungary | Debrecen | 2009/10.01 | == | <u>4 weeks</u> | 10–15 | (Molnár et al., 2010) |
| France | Paris | 2010.01–02 | == | == | 26.4 | (Lopez et al., 2013) |
| United Kingdom | London | 2020.06–07 | <u>12:00</u> | <u>30 min</u> | 17.3 \pm 3.0 | (Zazzeri et al., 2023) |
| United States | Los Angeles | 2006–2013 | <u>14:00</u> | <u>1 h</u> | 22.9 \pm 5.6 | (Newman et al., 2016) |
| United States | Los Angeles | 2014.11–2016.03 | <u>14:00</u> | <u>1 h</u> | 13.2 \pm 9.4 | (Miller et al., 2020) |
| United States | Indianapolis | 2010–2015 | <u>14:00</u> | <u>1 h</u> | 10.8 \pm 1.0 | (Turnbull et al., 2015) |
| China | Urumqi | 2014–2016 | <u>14:00-16:00</u> | <u>2 h</u> | 45.6 \pm 12.9 | (Zhou et al., 2020) |
| China | Lanzhou | 2014–2016 | <u>14:00-16:00</u> | <u>2 h</u> | 36.4 \pm 8.8 | (Zhou et al., 2020) |
| China | Xi'an | 2011–2013 | <u>afternoon</u> | == | 40.1 \pm 3.8 | (Zhou et al., 2022) |
| China | Xi'an | 2014–2016 | <u>afternoon</u> | == | 25.7 \pm 1.1 | (Zhou et al., 2022) |
| China | Suburban Xi'an | 2016. <u>01-11</u> | <u>14:00</u> | <u>15 min</u> | 23.5 \pm 6.5 | (Wang et al., 2018) |
| China | Near Xi'an | 2021.04–2022.03 | <u>14:00</u> | <u>40 min</u> | 13.1 \pm 10.9 | (Liu et al., 2024) |
| China | Beijing | 2014. <u>01-12</u> | <u>10:00</u> | <u>10 min</u> | 39.7 \pm 36.1 | (Niu et al., 2016) |
| China | Beijing | 2014–2016 (<u>Jan</u>) | <u>14:00-16:00</u> | <u>2 h</u> | 27.0 \pm 0.3 | (Zhou et al., 2020) |
| China | Beijing | 2020 winter | <u>14:00</u> | <u>1 h</u> | 19.7 \pm 22.0 | (Wang et al., 2022b) |
| China | Wuhan | 2014–2016 | <u>14:00-16:00</u> | <u>2 h</u> | 34.5 \pm 10.0 | (Zhou et al., 2020) |
| China | Xiamen | 2014 | <u>10:00</u> | <u>10 min</u> | 13.6 \pm 12.3 | (Niu et al., 2016) |
| China | Guangzhou | 2011 | <u>20:00</u> | <u>45 min</u> | 23.7 \pm 12.9 | (Ding et al., 2013) |
| China | Guangzhou | 2022 | <u>13:00-17:00</u> | <u>20 min</u> | 15.3 \pm 5.2 | this study |
| China | Shenzhen | 2022 | <u>13:00-17:00</u> | <u>20 min</u> | 13.7 \pm 10.2 | this study |
| China | Zhanjiang | 2022 | <u>13:00-17:00</u> | <u>20 min</u> | 10.0 \pm 5.2 | this study |
| China | Shaoguan | 2022 | <u>13:00-17:00</u> | <u>20 min</u> | 8.2 \pm 7.0 | this study |

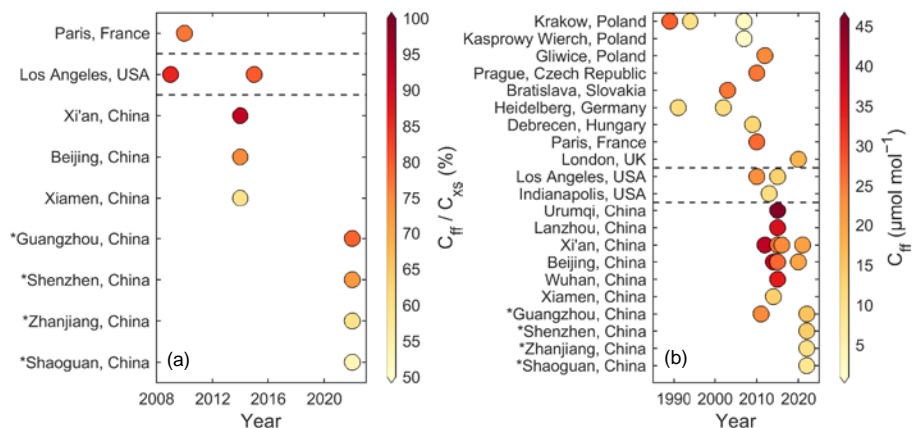


Figure E1 Comparison of (a) C_{fr} fractions in C_{xs} and (b) C_{fr} concentrations derived from $\Delta(^{14}\text{CO}_2)$ measurements from previous studies and this study (*) in various cities across European countries (Kuc et al., 2003; Levin and Rödenbeck, 2008; Molnár et al., 2010; Svetlik et al., 2010; Zimnoch et al., 2012; Lopez et al., 2013; Piotrowska et al., 2020; Zazzeri et al., 2023), United States (Turnbull et al., 2015; Newman et al., 2016; Miller et al., 2020), and China (Ding et al., 2013; Niu et al., 2016; Zhou et al., 2020; Wang et al., 2022b; Liu et al., 2024; Zhou et al., 2022). In (a), Los Angeles, Beijing, Guangzhou, and Shenzhen are megacities, Xi'an is a supercity, and other are large and medium cities (Council, 2022). Values in (a) and (b) refer to Tables E1 and E2, respectively.

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Appendix F: HYSPLIT back trajectories and FLEXPART footprints

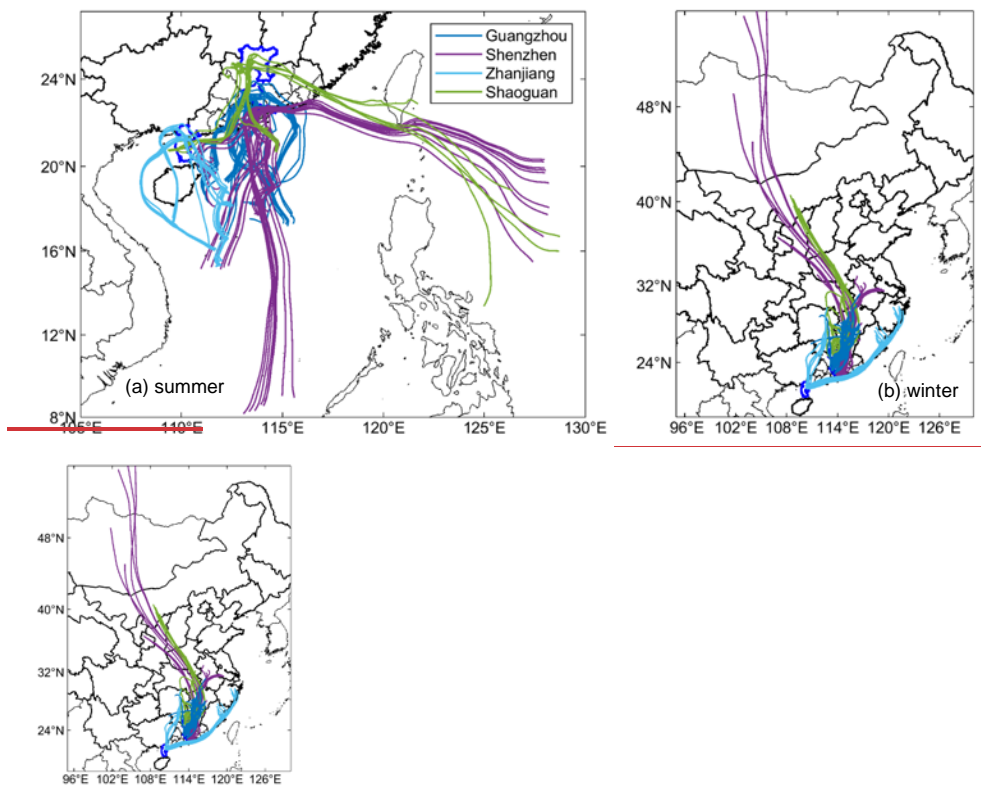


Figure F1 HYSPLIT back trajectories for a 72-h duration for Guangzhou, Shenzhen, Zhanjiang, and Shaoguan in (a) summer and (b) winter. The trajectories reveal maritime inflow in summer and continental inflow in winter, corroborating the ERA5-based analysis. National boundaries were taken from Natural Earth (<https://www.naturalearthdata.com/>, last accessed: 9 March 2024).

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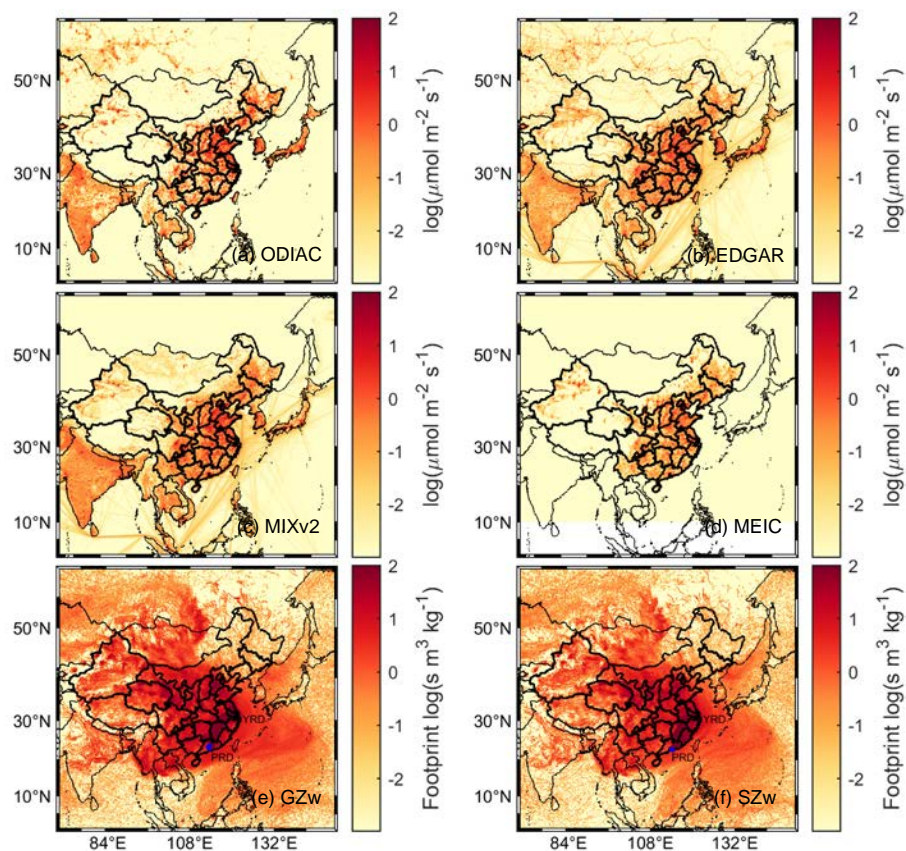


Figure F2 Annual mean C_{ff} emissions in China from four gridded inventory datasets: (a) ODIAC, (b) EDGAR, (c) MIXv2, and (d) MEIC, and FLEXPART footprints simulating C_{ff} emissions for (e) Guangzhou (GZ), (f) Shenzhen (SZ) in winter (w) by releasing particles at the blue sampling sites at heights from 0–100 m a.s.l. over a period of 30 days. The YRD and PRD represent Yangtze River Delta and Pearl River Delta urban agglomeration labelled in (e) and (f). Boundaries of nations and Chinese provinces were obtained from Natural Earth (<https://www.naturalearthdata.com/>, last accessed: 9 March 2024).

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Appendix G: Transport representativeness analysis

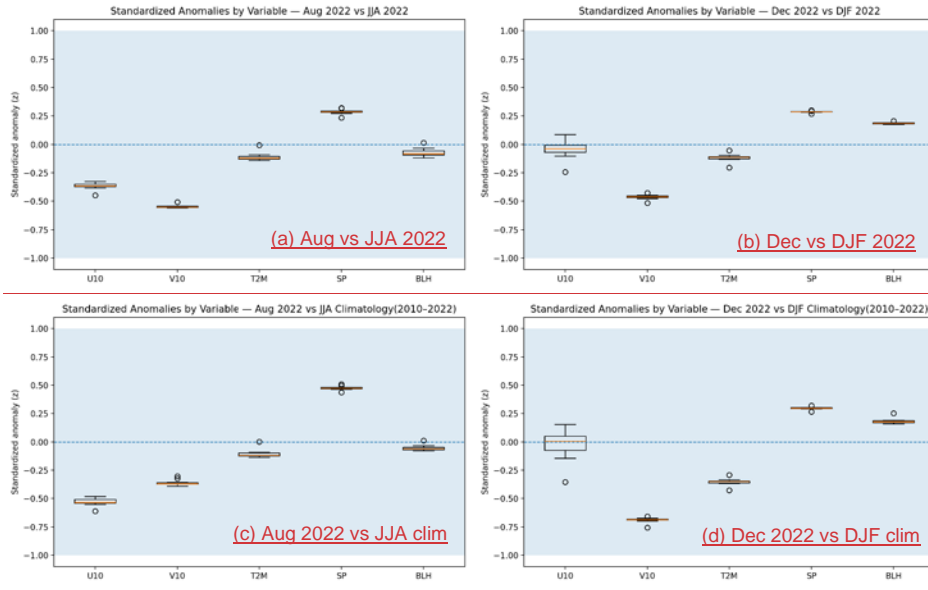
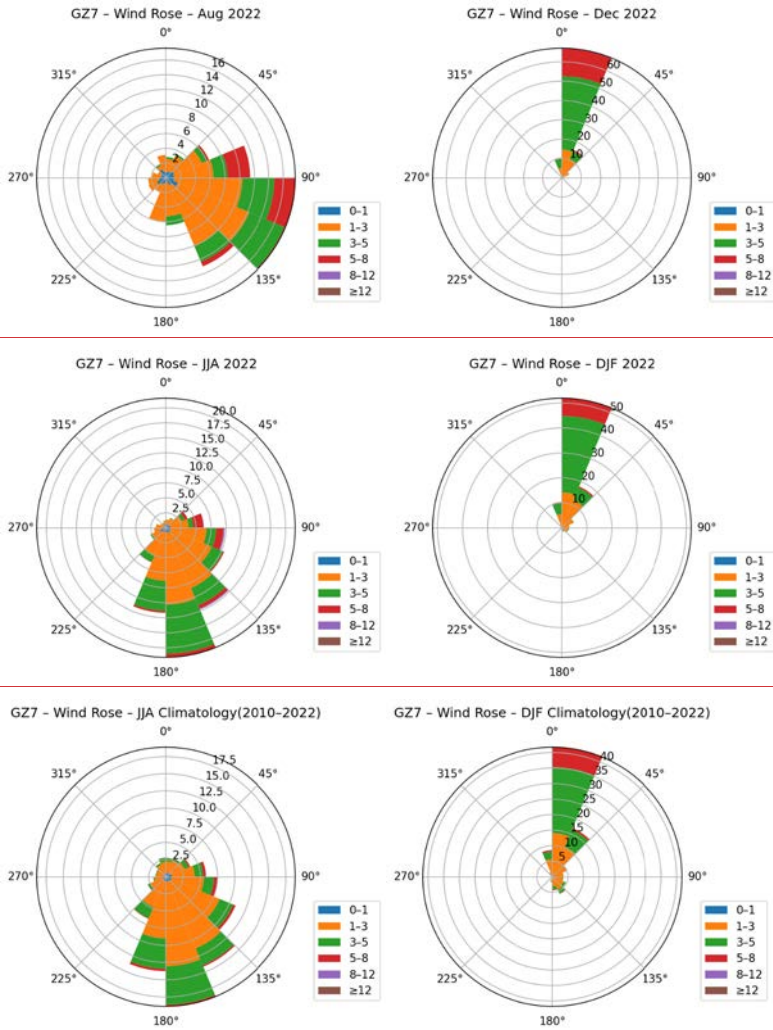


Figure G1 Box-and-whisker plots of standardized anomalies (z) by ERA5 meteorological variables (U10, V10, T2M, SP, BLH) across Guangzhou sites (GZ1-10) for (a) Aug 2022 vs JJA 2022, (b) Dec 2022 vs DJF 2022, (c) Aug 2022 vs JJA climatology (2010-2022), and (d) Dec 2022 vs DJF climatology (2010-2022). U10, V10, T2M, SP, BLH are 10 m zonal and meridional winds (U10, V10), 2 m temperature (T2M), surface pressure (SP), and planetary boundary-layer height (PBLH), respectively. The shaded region denotes $|z| < 1$ (typical range).

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Figure G2 ERA5 wind roses for GZ7 site (wind speed unit: m s^{-1}) showing six panels: (a) August 2022, (b) December 2022, (c) JJA 2022, (d) DJF 2022, (e) JJA climatology (2010–2021), and (f) DJF climatology (2010–2021). These illustrate that both August and December are consistent with their canonical summer and winter flow regimes.

Appendix H: Historical comparison and corrections

Table H1. Summary of all available C_{ff} datasets for historical variations used in this study and referenced from previous literature, including site type, coordinates, sampling period, time, number of samples (n), and references.

| City | Site type | Site location | Sampling period | Sampling time | Duration | n | Reference |
|-----------|------------------|--|---------------------|---------------|----------|-------|----------------------|
| Guangzhou | Urban | GZ7 in Fig. 1f | Oct 2010 – Nov 2011 | 20:00 | 45 min | 58 | (Ding et al., 2013) |
| | Urban & suburban | GZ1–10 in Figs. 1ef | Aug/Dec 2022 | 13:00–17:00 | 20 min | 40/40 | This study |
| Beijing | Urban | RCEES ^a (40.02°N, 116.34°E, 15 m a.g.l.) | Jan–Dec 2014 | 10:00 | 10 min | 24 | (Niu et al., 2016) |
| | | | Jan/Jul 2014–2016 | 14:00–16:00 | 2 h | 42/42 | (Zhou et al., 2020) |
| | | | Dec 2020–Jan 2021 | 14:00 | 1 h | 31 | (Wang et al., 2022b) |
| Xi'an | Urban | IEECAS ^b (34.23°N, 108.89°E, 10 m a.g.l.) | 2011–2013 | afternoon | -- | ~120 | (Zhou et al., 2022) |
| | | | 2014–2016 | afternoon | -- | ~75 | |
| | Suburban | two sites (34.34°N, 108.86°E; 34.21°N, 108.88°E) | Jan–Nov 2016 | 14:00 | 15 min | 38 | (Wang et al., 2018) |
| | Suburban | Qinling Mountain (34.06°N, 108.34°E) | Apr 2021–Mar 2022 | 14:00 | 40 min | 24 | (Liu et al., 2024) |

^a Research Center for Eco-Environmental Sciences, Chinese Academy of Sciences (RCEES). ^b Institute of Earth Environment, Chinese Academy of Sciences (IEECAS).

Table H2. Harmonized comparison of C_{ff} mole fractions at identical sites, seasons, and sampling times, after correction to common background conditions.

| City | Site type | Site location | Sampling period | Sampling time | Duration | n | Background | C_{ff} ($\mu\text{mol mol}^{-1}$) | Corrected C_{ff} ($\mu\text{mol mol}^{-1}$) |
|-----------|-----------|---------------------------------------|------------------|---------------|----------|-----|-----------------------------|---------------------------------------|---|
| Guangzhou | Urban | GZ7 (GIGCAS) ^a | Winter 2010 | 20:00 | 45 min | 8 | remote regions ^b | 28.4 ± 15.0 | 21.6 ± 16.9 ^d |
| | | | Winter 2022 | 13:00–17:00 | 20 min | 4 | NL air | 16.8 ± 3.4 | 11.6 ± 3.4 ^e |
| Beijing | Urban | RCEES (40.02°N, 116.34°, 15 m.a.g.l.) | Winter 2014–2016 | 14:00–16:00 | 2 h | 21 | QXL | 27.0 ± 0.3 | 27.6 ± 0.3 ^f |
| | | | Winter 2020 | 14:00 | 1 h | 31 | WLG ^c | 19.7 ± 22.0 | 19.7 ± 22.0 |

| | | | | | | | | | |
|-------|----------|--|-----------|-----------|--------|------|-----|-------------|-------------|
| Xi'an | Urban | IEECAS (34.23°N, 108.89°E, 10 m.a.g.l.) | 2011–2013 | afternoon | -- | ~120 | WLG | 40.1 ± 3.8 | 40.1 ± 3.8 |
| | | | 2014–2016 | afternoon | -- | ~75 | WLG | 25.7 ± 1.1 | 25.7 ± 1.1 |
| | Suburban | two sites (34.34°N, 108.86°E; 34.21°N, 108.88°E) | 2016 | 14:00 | 15 min | 38 | WLG | 23.5 ± 6.5 | 23.5 ± 6.5 |
| | Suburban | Qinling Mountain (34.06°N, 108.34°E) | 2021–2022 | 14:00 | 40 min | 24 | WLG | 13.1 ± 10.9 | 13.1 ± 10.9 |

^a Guangzhou Institute of Geochemistry, Chinese Academy of Sciences (GIGCAS). ^b remote regions (Qinghai, Gansu, and Tibet). ^c C_{ff} concentrations in winter Beijing obtained from Wang et al.⁴⁰ were estimated based on the background atmospheric $\delta(^{14}CO_2)$ measurements from Waliguan (Liu et al., 2024). ^d corrected to 14:00 sampling and NL tree-ring background. ^e corrected to NL tree-ring background. ^f corrected to the WLG background.

H1 Sampling-time difference (20:00 vs 14:00) in C_{ff} for Guangzhou. Ding et al. (2013) collected flask samples around 20:00 LT, immediately after the evening rush hour, whereas our 2022 samples were collected between 13:00 and 17:00 LT under well-mixed boundary-layer conditions. To quantify the potential diurnal C_{ff} bias, we used continuous CO observations near GZ7 from two independent sources: (1) the Guangdong Provincial Environmental Monitoring Centre and (2) the nationwide air quality database (<https://quotsoft.net/air/#messy>, last access: 18 October 2025), and applied the formulation

$$C_{ff} \approx \frac{\Delta CO}{R},$$

where $\Delta CO = CO - CO_{bg}$, and $R = \Delta CO / \Delta CO_{2,ff}$ denotes the emission ratio between CO and fossil-fuel CO_2 . The value of R is strongly time-dependent and source-specific. Nighttime R values tend to be higher than daytime values because (i) nighttime emissions are dominated by direct fossil-fuel combustion while biogenic CO_2 sources (e.g., respiration) remain constant but emit no CO, and (ii) oxidative sinks (e.g., OH radicals) are weaker at night. Therefore, applying an afternoon-derived R to nighttime data likely provides a lower bound for the actual nighttime C_{ff} enhancement.

Scheme 1 (this study's observation, December 2022):

We used the December diurnal mean CO data at a site close to GZ7, subtracted the NL background to obtain ΔCO , and divided by the afternoon-specific $R = 13.3 \text{ ppb ppm}^{-1}$ (derived from the regression between ΔCO and ^{14}C -based $\Delta CO_{2,ff}$). The ΔCO increased from 168.3 ppb at 14:00 to 220.7 ppb at 20:00, corresponding to an estimated nighttime C_{ff} enhancement of approximately 3.2 ppm, or about 21 % higher than the afternoon value. Because the slope R was determined during well-mixed afternoon periods, this result likely represents a lower limit; the actual nighttime–afternoon C_{ff} contrast may be smaller.

Scheme 2 (Guangzhou dataset, winter 2023, supplementary analysis):

We applied the same approach to the continuous CO data at the site close to GZ7 for the winter season (December 2023 – February 2024). After subtracting the NL background, ΔCO was divided by the seasonal $R = 9.08 \text{ ppb ppm}^{-1}$ obtained from regressions of CO against total CO₂ (Zhang et al., 2025). As this R reflects bulk CO₂ rather than specifically fossil-fuel CO₂, we applied an empirical correction (dividing R by 0.8; Turnbull et al. (2011)). The resulting analysis indicates that ΔCO increased by 66.9 ppb from 14:00 to 20:00, implying a C_{ff} enhancement difference of roughly 5.9 ppm ($\approx 35\%$).

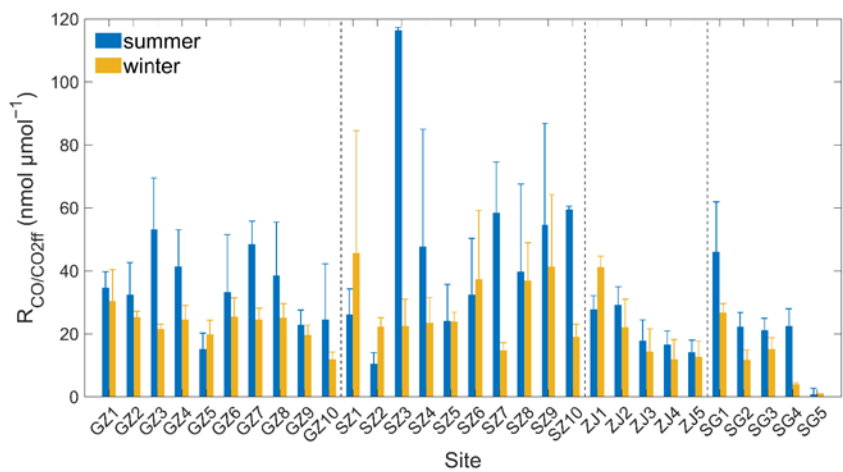
Overall, while continuous CO_{2ff} data are not available for 2022, our CO-based analysis suggests that the C_{ff} signal at 20:00 is moderately higher than that at 14:00. This finding is consistent with the reviewer's expectation that post-rush-hour conditions retain a stronger fossil-fuel signature compared with the well-mixed afternoon atmosphere. The semi-quantitative assessment indicates that the C_{ff} concentration during post-rush-hour (20:00) is approximately 21–35 % higher than during well-mixed afternoon periods, consistent with the expected diurnal accumulation of fossil-fuel CO₂ under weaker nighttime boundary-layer mixing.

Appendix IG: Comparison of contributions of coal, oil and natural gas to C_{ff} concentrations in various cities

Table IG1 Comparison of contributions of coal, oil and natural gas to C_{ff} concentrations in various cities

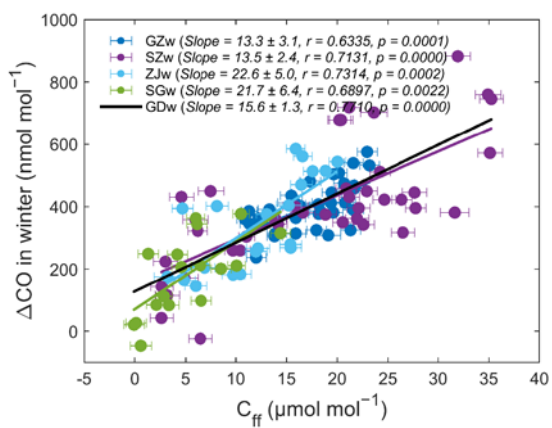
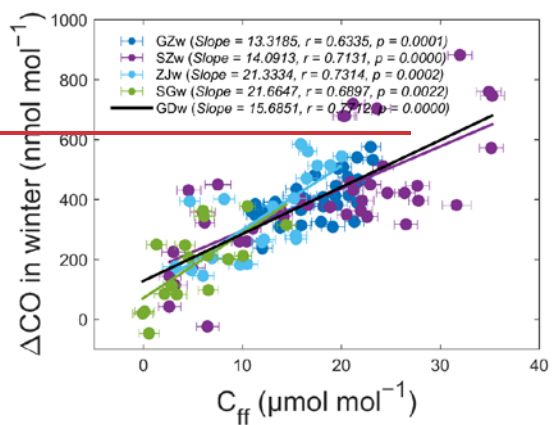
| City | Time | F_{coal} (%) | F_{oil} (%) | F_{ng} (%) | References |
|-------------|------------------|-----------------------|----------------------|---------------------|-------------------------|
| Paris | 2010 | < 1 | 30 | 70 | (Lopez et al., 2013) |
| Los Angeles | 2007.10 | < 1 | 69 | 31 | (Djuricin et al., 2010) |
| Los Angeles | 2007.12 | < 1 | 61 | 39 | (Djuricin et al., 2010) |
| Los Angeles | 2008.02 | < 1 | 58 | 42 | (Djuricin et al., 2010) |
| Los Angeles | 2008.04 | < 1 | 52 | 48 | (Djuricin et al., 2010) |
| Los Angeles | 2006-2013 winter | < 1 | 68 | 32 | (Newman et al., 2016) |
| Los Angeles | 2006-2013 summer | < 1 | 55 | 45 | (Newman et al., 2016) |
| Xi'an | 2014 winter | 72.6 ± 10.4 | 13.8 ± 10.4 | 13.6 | (Zhou et al., 2014) |
| Xi'an | 2019.12-2020.01 | 54 ± 4 | 24 ± 14 | 22 ± 13 | (Wang et al., 2022b) |
| Beijing | 2020.12-2021.01 | 17 ± 10 | 28 ± 19 | 55 ± 9 | (Wang et al., 2022b) |
| Guangzhou | 2022 winter | 49 ± 25 | 29 ± 22 | 22 ± 19 | this study |
| Shenzhen | 2022 winter | 47 ± 25 | 29 ± 21 | 24 ± 20 | this study |
| Zhanjiang | 2022 winter | 43 ± 24 | 29 ± 21 | 28 ± 21 | this study |
| Shaoguan | 2022 winter | 39 ± 24 | 34 ± 23 | 27 ± 21 | this study |

Appendix JH: $R_{CO/CO2ff}$ for sites, cities and comparison



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Figure JH1 $R_{CO/CO2ff}$ averages at the 30 sampling sites



945 **Figure JH2** ΔCO : C_{ff} for Guangzhou, Shenzhen, Zhanjiang, and Shaoguan in winter

Table JH1 Observational emission ratios of CO to C_{ff} (R_{CO/CO2ff}) for China and Chinese cities

| City | Time | R _{CO/CO2ff} (nmol μmol ⁻¹) | Method ^a | References |
|------------------------|------------------|--|---------------------|-------------------------------|
| China | 2001 | 68.8 | I | (Suntharalingam et al., 2004) |
| China | 2004–2010 | 44 ± 3 | II | (Turnbull et al., 2011) |
| China | 1998 | 56.3 | I | (Tohjima et al., 2014) |
| China | 2010 | 37.5 | I | (Tohjima et al., 2014) |
| Mainland China | 2009 | 36.3 | I | (Fu et al., 2015) |
| Yellow Sea | 2016 | 35.0 | I | (Tang et al., 2018) |
| China, CN ^b | 2014–2016 | 31 ± 8 | II | (Lee et al., 2020) |
| China, CE ^b | 2014–2016 | 36 ± 2 | II | (Lee et al., 2020) |
| China, CB ^b | 2014–2016 | 29 ± 8 | II | (Lee et al., 2020) |
| China, OB ^b | 2014–2016 | 31 ± 4 | II | (Lee et al., 2020) |
| Beijing | 2005 | 54.3 | I | (Han et al., 2009) |
| Beijing/Tianjin | 2009–2010 | 59.5 | I | (Silva et al., 2013) |
| Beijing | 2014 | 30.4 ± 1.6 | II | (Niu et al., 2018) |
| Rural Beijing | 2004 | 72.3 | I | (Wang et al., 2010) |
| Rural Beijing | 2005 | 52.5 | I | (Wang et al., 2010) |
| Rural Beijing | 2006 | 48.1 | I | (Wang et al., 2010) |
| Rural Beijing | 2007 | 43.7 | I | (Wang et al., 2010) |
| Rural Beijing | 2008 | 47.0 | I | (Wang et al., 2010) |
| Rural Beijing | 2009–2010 | 47 ± 2 | II | (Turnbull et al., 2011) |
| Xi'an | 2016 | 46 ± 13 | II | (Wang et al., 2018) |
| Near Xi'an | 2021 | 23 ± 6 | II | (Liu et al., 2024) |
| Xiamen | 2014 | 29.6 ± 3 | II | (Niu et al., 2018) |
| Guangzhou | 2009–2010 | 35.8 | I | (Silva et al., 2013) |
| Guangzhou | 2014–2017 winter | 23.8 | I | (Mai et al., 2021) |
| Guangzhou | 2022 winter | 13.3 ± 3.10.00002 | II | Fig. JH2, this study |
| Shenzhen | 2022 winter | 13.54.1 ± 2.40.000003 | II | Fig. JH2, this study |
| Zhanjiang | 2022 winter | 22.64.3 ± 5.00.00002 | II | Fig. JH2, this study |
| Shaoguan | 2022 winter | 21.7 ± 60.00004 | II | Fig. JH2, this study |

^a by correction from R_{CO/CO2} by increased 20 % (Method I) and estimation from Δ¹⁴C measurements (Method II), ^b CN represents the air masses from northeast China, CE for central eastern China around the Shandong area, CB for continental

950 background air, and OB for ocean background.

Code availability

The FLEXPART 10.4 model is available at <https://www.flexpart.eu/>. The MixSIAR 3.1.12 model is available via GitHub at <https://brianstock.github.io/MixSIAR/index.html>. The HYSPLIT model is available at <https://www.arl.noaa.gov/hysplit/>. In this study, commercial software such as MATLAB R2023a, and public software such as R 4.3.2 and Python 3.9 are used for data processing and result visualization.

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Data availability

Data generated in this study are available in Supplement Dataset 1 and seasonal averages in Table A1. Additional data related to this paper may be requested from the corresponding authors. The Carnegie Ames Stanford Approach Global Fire Emissions Database Version 4 (CASA-GFED4s) dataset is available at https://daac.ornl.gov/VEGETATION/guides/fire_emissions_v4_R1.html. The CASA-GFED3 dataset is available at <http://nacp-files.nacarbon.org/nacp-kawa-01/>. The Open-source Data Inventory for Anthropogenic CO₂ (ODIAC) is available from <https://db.cger.nies.go.jp/dataset/ODIAC/>. The Emissions Database for Global Atmospheric Research (EDGAR) Global Greenhouse Gas and Air Pollutant Emissions are from <https://edgar.jrc.ec.europa.eu>. The Multi-resolution Emission Inventory for China (MEIC) and the Open Biomass Burning Emission Inventory for China (OBBEIC) are available from <http://meicmodel.org.cn>. The MIXv2 Asian emission inventory (MIXv2) is available from <https://csl.noaa.gov/groups/csl4/modeldata/data/Li2023/>. The National Centers for Environmental Prediction's Climate Forecast System (CFSv2) Reanalysis data that drive the FLEXPART model is available at <https://rda.ucar.edu/datasets/ds094.0/>. The National Centers for Environmental Prediction's Global Data Assimilation System (GDAS) Reanalysis data that drives the HYSPLIT model is available at <ftp.arl.noaa.gov/pub/archives/reanalysis>.

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970 Author contribution

G.Z., D.C., Jun Li, and P.L. conceived and designed the study. Almost all authors participated in the sampling organized by Jun Li and P.L.. Z.N. provided data in Beijing and Xi'an. P.D. provided data from Ding et al., (2013) in Guangzhou. R.L. conducted the sample graphitization. Sanyuan Z. handled the ¹⁴C measurement by AMS. P.L. and B.L. performed the simulations. P.L. conducted the data search and analysis, and wrote the article, with contributions from G.Z., Jun Li, Z.C., Jing L., and T.Z. for revisions and improvements.

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Competing interests

The authors declare no competing interest.

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