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RESEARCH ARTICLE OPEN ACCESS

Can Common Recycling Targets Deliver Convergence? Evidence From EU Municipal Waste Recycling, 2004–2023

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Received: 25 March 2026 | **Revised:** 5 August 2026 | **Accepted:** 1 September 2026

Keywords: convergence | EU-27 | lock-in | municipal waste recycling | national sword | waste framework directive

ABSTRACT

The EU Waste Framework Directive sets common municipal waste recycling targets for all Member States, yet large cross-country gaps persist. This paper examines whether EU-27 recycling rates converged over 2004–2023, whether convergence stalled after 2018, and which factors shaped catch-up. Using an EU-27 panel, we combine sigma-convergence, beta-convergence, and the Phillips–Sul log-t test. Recycling performance became markedly less dispersed over the full period: the coefficient of variation fell from 0.810 in 2004 to 0.380 in 2023, the absolute beta-convergence coefficient was −0.040 ($p < 0.001$), and the Phillips–Sul test did not reject full-panel convergence. However, convergence stalled in 2018. A Chow test identifies a structural break, while placebo tests show that the dispersion trend shifted from negative to near zero in that year. Using bilateral UN Comtrade data on pre-ban exports of recovered paper and waste plastics to China, we test whether post-2018 convergence varied with exposure to the Chinese import ban. Conditional beta-convergence estimates show that the WFD compliance gap is strongly associated with catch-up, but the relationship is nonlinear: countries with the largest gaps improve more slowly. This pattern is consistent with structural lock-in. Greater landfill dependence was also associated with lower ESIF absorption rates, indicating co-located structural and institutional barriers. As of 2023, only 6 of 27 Member States had reached the 2025 recycling target of 55%. These findings suggest that common targets support convergence, although the process remains vulnerable to external shocks and structural and institutional constraints.

1 | Introduction

Municipal waste management and recycling in the European Union constitute a critical domain in the transition toward a circular economy (European Commission 2020; Seyyedi et al. 2024). This domain is directly linked to the coverage of separate collection systems, treatment capacity, and the absorptive capacity of secondary material markets. It is implemented primarily by local governments and public service systems (Albizzati et al. 2024). Although municipal waste accounts for only a limited share of total waste generated in the EU, it is characterised by complex composition, high management difficulty, and strong public visibility. It therefore occupies a prominent position in policy evaluation and public accountability (D'Adamo et al. 2024).

The EU *Waste Framework Directive* (WFD) establishes a legally binding common target system for Member States (European Commission 2025). Directive 2008/98/EC set out the fundamental institutional framework for waste management, including the waste hierarchy and extended producer responsibility, and introduced a target of at least 50% for preparing for re-use and recycling by 2020 (European Commission 2008). Subsequently, Directive (EU) 2018/851, adopted under the Circular Economy Package, further raised the municipal waste recycling targets to 55% by 2025, 60% by 2030, and 65% by 2035 (European Commission 2018). Yet Member States entered this common target system with markedly different waste-management structures, including varying levels of landfill dependence, separate collection coverage, treatment capacity, and secondary-material market development. Despite common rules and shared targets, substantial cross-country differences in recycling performance have therefore persisted over time. This raises a central governance question: can common targets induce systematic convergence in Member State performance, or will pre-existing structural disparities continue to persist and produce a stratified long-term recycling pattern?

This question matters because convergence is not only a statistical property of recycling rates, but also a test of whether common EU targets can translate into comparable progress across heterogeneous national waste-management systems. First, the design of WFD targets implicitly assumes that Member States possess the capacity to achieve them (Möslinger et al. 2023). If convergence is absent, the continued use of uniform threshold-based targets may generate persistent compliance pressure and governance frustration, thereby undermining policy implementability and credibility. Second, convergence analysis can provide a clearer framework for identifying gaps in infrastructure investment and capacity building. Which countries, or which mechanisms, determine whether the EU as a whole can meet its targets on time has direct implications for the prioritisation of funding allocation, technology choices, and regulatory resources (Hondroyiannis et al. 2024; Tomić and Schneider 2020). Third, from a waste-management perspective, the recycling rate is not only an indicator of environmental performance but also a composite measure of the operational performance of waste management systems. Although it does not capture all dimensions of circular economy transition, it is the statutory outcome indicator most directly linked to the WFD municipal waste targets and is consistently reported across Member States over time (Salmenperä et al. 2021). Because recycling performance depends on separate collection, treatment and biological recovery capacity, secondary material markets, and cross-border trade, it provides a focused and policy-relevant indicator for examining whether cross-country convergence occurred under common EU recycling targets (Albizzati et al. 2024; Salmenperä et al. 2021).

At the same time, the evolution of EU recycling performance has not occurred within a stable institutional and market environment. Since 2018, China's National Sword policy has imposed import restrictions that have substantially altered the cross-border flows and price structures of recyclable materials worldwide (Brooks et al. 2018). A large body of research based on trade data has shown that the Chinese ban generated clear cascading effects on global plastic waste trade, triggering trade diversion and pressure for domestic disposal restructuring across countries (Tran et al. 2021; Wen et al. 2021). For economies that had long relied on exports to absorb low-value or mixed recyclables, such external shocks may have weakened the economic viability of recycling systems through market channels, thereby affecting their convergence paths.

In addition, the statistical accounting rules under the WFD became more stringent after the 2018 revision. Commission Implementing Decision (EU) 2019/1004 introduced more harmonised requirements for the calculation, verification, and reporting of waste data, in line with the principle of measuring waste at the point at which it enters the final recycling operation (European Commission 2019a). These implementation arrangements make the period around 2020 an important turning point, both because statistical definitions became stricter and because the comparability of reported data requires renewed scrutiny. For a long panel spanning 2004 to 2023, this creates a methodological challenge that must be addressed through robustness checks. It also forms part of the policy reality because stricter accounting rules may themselves affect compliance assessments and policy incentives.

Against this background, this study focuses on municipal waste recycling rates in the EU-27 from 2004 to 2023 and addresses three interrelated questions. First, do Member States display a long-run convergence trend in municipal waste recycling rates? Second, did this process stall after 2018? Third, if convergence weakened, what roles were played by external shock exposure and domestic structural and institutional constraints? To answer these questions, this study combines sigma-convergence, absolute and conditional beta-convergence, and the Phillips and Sul (2007) log-*t* test to assess changes in dispersion, catch-up speed, and heterogeneous convergence paths.

This study extends the literature in three ways. First, it uses formal structural break tests and placebo-year scans to assess whether 2018 marks a substantively relevant break in the convergence process. Second, it constructs a country-level indicator of exposure to China from bilateral UN Comtrade exports of recovered paper and waste plastics during 2013–2017 to assess whether post-2018 convergence patterns differed with pre-ban exposure. Third, it operationalises WFD target pressure as a compliance gap and allows for nonlinear effects, allowing the estimated catch-up relationship to vary with the size of the compliance gap. It then combines historical landfill dependence with ESIF absorption capacity to examine whether structural and institutional barriers are co-located across Member States.

2 | Literature Review

2.1 | EU Waste Policy, Circular Economy, and Uniform Recycling Targets

EU waste policy has gradually evolved from disposal control toward circular-economy governance. The Waste Framework Directive established the waste hierarchy, clarified Member State obligations, and provided the legal basis for prevention, preparing for re-use, recycling, and recovery (European Commission 2008). Its 2018 amendment further raised municipal waste recycling targets and strengthened separate collection and circular-economy requirements (European Commission 2018). At the same time, the 2015 and 2020 Circular Economy Action Plans broadened the policy agenda from waste treatment to product design, resource efficiency, secondary raw materials, priority value chains, and waste-export governance (European Commission 2020; 2015). This policy evolution makes municipal recycling rates a central statutory performance indicator, while also showing that recycling is only one dimension of a broader circular-economy transition.

A key unresolved issue in this policy framework is whether uniform targets can produce comparable outcomes across highly heterogeneous national systems. Previous policy assessments show that landfill diversion and recycling improvement depend on combinations of regulatory pressure, economic instruments, infrastructure investment, and adaptation to local conditions (European Environment Agency 2009). Extended producer responsibility can also support recycling only when responsibilities, cost allocation, monitoring, and incentives are well designed (OECD 2016). Therefore, common EU targets create a shared direction and compliance pressure, but their effects may differ across Member States with different landfill dependence, administrative capacity, local implementation systems, and secondary-material markets. This tension between common targets and heterogeneous capacity provides the central policy background for the present study.

The interpretation of recycling-rate convergence also depends on data comparability. Commission Implementing Decision (EU) 2019/1004 harmonised rules for calculating, verifying, and reporting waste data, including the principle of measuring waste entering final recycling operations (European Commission 2019a). For long-period panel analysis, this means that recycling rates are not merely technical indicators; common statistical rules and evolving EU reporting requirements also shape them. This is why the present study treats recycling performance as a policy-facing indicator embedded in EU target-setting, accounting standards, and implementation capacity.

2.2 | Waste-Management Performance and Circular-Economy Implementation

The academic literature increasingly treats waste management as an operational domain of the circular economy rather than a purely downstream disposal issue. Wilson (2023) connects current circular-economy debates with the historical development of integrated sustainable waste management. Salmenperä et al. (2021) identify institutional, market, technological, and behavioural conditions as critical factors for enhancing circularity in waste management. Tomić and Schneider (2020) further show that changes in waste-management system structure can generate socio-economic effects, indicating that recycling policies affect not only environmental outcomes but also cost distribution and local system organisation.

Recent empirical studies similarly show that municipal waste performance is shaped by system design and socioeconomic conditions. Albizzati et al. (2024) model the environmental and economic impacts of municipal waste management in Europe, while D'Adamo et al. (2024) link municipal waste efficiency to EU sustainability and circular-economy performance. Hondroyiannis et al. (2024) show that recycling-rate performance is associated with socioeconomic determinants across EU countries and regions. Assessment-oriented work also stresses that conclusions depend on indicators, system boundaries, and evaluation methods (Campitelli et al. 2022). Together, these studies justify using municipal waste recycling rates as a comparable and policy-relevant outcome, while also making clear that recycling convergence should be interpreted within broader waste-system and circular-economy contexts.

EU waste governance has also become increasingly waste-stream-specific. The waste electrical and electronic equipment (WEEE) Directive established a dedicated framework for electronic waste collection, recovery, and recycling (European Commission 2012); the Single-Use Plastics Directive introduced targeted measures for plastic products, including reduction, labelling, extended producer responsibility (EPR), and collection requirements (European Commission 2019b); the Batteries Regulation integrated sustainability, collection, recycling, reuse, and information requirements across the battery value chain (European Commission 2023); and the Packaging and Packaging Waste Regulation further strengthened packaging governance through prevention, recyclability, recycled-content, and packaging-waste rules. These regulations show that municipal recycling performance is influenced not only by general targets but also by product design, producer responsibility, material composition, and specialised recovery systems. They also explain why a single recycling-rate target can provide a common benchmark but cannot capture all dimensions of waste-system transformation.

2.3 | Convergence Research and EU Waste-Performance Differences

Convergence analysis originates from macroeconomic growth research and examines whether lagging units catch up with leading ones. Barro and Xavier (1992) formalised beta-convergence as a negative relationship between initial level and subsequent growth, while sigma-convergence evaluates whether cross-sectional dispersion declines over time. Phillips and Sul (2007) proposed the log-*t* test and a time-varying factor framework that allows heterogeneous transition paths and convergence clubs. Although convergence methods are widely used in environmental and energy economics, their application to waste-management performance remains comparatively limited.

The studies closest to this paper indicate that EU waste performance is heterogeneous and that catch-up is not automatic. Marin et al. (2018) examine catching-up in EU waste management and highlight the role of policy stringency, technological capacity, and national heterogeneity. Castillo-Giménez et al. (2019) show that Member States differ substantially in municipal waste treatment performance. Sastre et al. (2018) further demonstrate that distance to EU recycling targets can vary strongly at the regional level, suggesting that national averages may conceal important sub-national implementation gaps. These studies support the use of convergence analysis in the waste field, but they leave open whether the post-2018 period altered the convergence path of municipal recycling rates.

This gap is important because the recent EU waste-policy environment differs from the period covered by much of the earlier literature. Since 2018, the EU has faced both stricter circular-economy targets and stronger external pressure from disruptions in recyclable-material trade. Existing studies have not yet systematically examined whether EU-27 municipal waste recycling rates converged over the full 2004–2023 period, whether convergence stalled after 2018, or whether the speed of catch-up differed according to exposure to external shocks and domestic structural constraints. The present study addresses this gap by combining sigma-convergence, beta-convergence, and the Phillips–Sul log-*t* test with structural break and mechanism analysis.

2.4 | External Shocks, Cross-Border Waste Flows, and Secondary-Material Markets

EU recycling systems are embedded in global secondary-material markets. Before 2018, exports of recovered paper and waste plastics to China provided an important outlet for recyclable materials from many high-income economies. Brooks et al. (2018) identify China's import ban as a major disruption to global plastic waste trade, while Wen et al. (2021) show that the ban reshaped global plastic-waste flows and redistributed environmental impacts. Tran et al. (2021) further discuss how China's tightening environmental regulations affected international waste trade and logistics. These studies indicate that domestic recycling performance can depend on external market access, especially for lower-value or mixed recyclable materials.

For the EU, the National Sword shock raises a convergence question rather than only a trade question. Member States differed in their pre-ban exposure to Chinese demand for recovered paper and waste plastics. Countries more dependent on external outlets may have experienced stronger disruption when export markets contracted, while countries with stronger domestic or intra-European treatment capacity may have been less vulnerable. At the same time, the EU has strengthened controls on waste exports and shipments, reflecting a policy shift toward reducing externalisation and strengthening internal circular-material markets (European Commission 2024). The present study contributes to this literature by constructing a country-level exposure to China indicator and testing whether post-2018 convergence dynamics differed between high- and low-exposure Member States.

2.5 | Implementation Capacity, Structural Lock-In, and Research Gap

The effectiveness of common recycling targets ultimately depends on implementation capacity. Local collection systems, separate collection coverage, route design, treatment contracts, household participation, monitoring, and charging schemes all influence recycling outcomes. Hannan et al. (2020) review solid-waste collection optimisation and show that collection design, routing constraints, logistics management, cost, fuel use, labour requirements, and emissions are central operational dimensions of municipal waste-system performance. Romano and Masserini (2023) further show that pay-as-you-throw tariffs can reduce total and unsorted urban waste while improving the quantity and quality of separate collection, indicating that charging schemes and household-level incentives can directly affect recycling-related outcomes. From an EU implementation perspective, Chioatto et al. (2023) also show that the translation of EU waste legislation into municipal waste-management performance varies across regions, suggesting that national and regional implementation systems mediate the effects of common EU targets. Although these studies operate at different scales, they reinforce a common point: EU-level targets must be translated through national, regional, and local implementation systems before they can affect recycling performance.

Administrative capacity is particularly relevant in the EU context because waste-system upgrading often depends on cohesion-policy funding, project preparation, procurement, and regulatory compliance. Incaltarau et al. (2020) show that EU fund absorption varies across Member States and depends on administrative capacity and political governance. Dotti et al. (2024) and Incaltarau et al. (2020) conceptualise administrative capacity through absorption, compliance, and outputs, highlighting that policy implementation cannot be reduced to formal funding allocation. For waste management, this suggests that the countries most in need of recycling infrastructure may also face the greatest difficulty in absorbing and implementing EU-supported investments.

Waste-management systems also exhibit strong path dependence. Landfills, incinerators, sorting facilities, composting plants, anaerobic digestion units, and recycling facilities involve long-lived infrastructure, sunk costs, contracts, and institutional routines. Wilson (2023) and Campitelli et al. (2022) emphasise that waste systems evolve through historically embedded configurations, while Ntostoglou et al. (2025) show how technological and institutional lock-in can slow the transition toward sustainable biowaste valorisation. This lock-in perspective suggests that uniform recycling targets may generate nonlinear effects: moderate laggards may respond quickly to pressure from targets, whereas extreme laggards with high landfill dependence and weak institutional capacity may catch up more slowly.

Overall, the literature establishes that EU municipal waste recycling performance is shaped by common legal targets, circular-economy policy, waste-stream-specific regulation, local implementation systems, external recyclable-material markets, and structural lock-in. However, three issues remain insufficiently resolved. First, it is unclear whether EU-27 municipal waste recycling rates continued to converge over the full 2004–2023 period. Second, existing studies have not systematically tested whether convergence stalled after the 2018 external shock and associated policy-accounting changes. Third, limited evidence exists on how external exposure and domestic structural and institutional constraints are associated with catch-up. This study addresses these gaps by examining long-run convergence, identifying structural breaks, assessing the roles of exposure to China and WFD target pressure, and documenting the cross-country relationship between landfill dependence and ESIF absorption capacity.

Based on the preceding analysis, this study proposes the following hypotheses:

H1. *Municipal waste recycling rates in the EU-27 exhibited an overall convergence trend during* 2004–2023.

H2. *This convergence process experienced a structural break around* 2018, *after which convergence weakened significantly and may even have shifted to non-convergence or club divergence.*

H3. *Post-*2018 *convergence patterns differed between Member States with high and low pre-ban exposure to exports of recovered paper and waste plastics to China.*

H4. *The association between WFD compliance pressure and catch-up is nonlinear, with slower catch-up at larger compliance gaps. High landfill dependence and weak fund implementation and absorption capacity are expected to be co-located across lagging Member States.*

3 | Data and Empirical Strategy

3.1 | Data, Variables, and Descriptive Statistics

The primary variable is the municipal waste recycling rate for the EU-27 over 2004–2023, sourced from Eurostat. The recycling rate includes material recycling, composting, and anaerobic digestion, expressed as a percentage of municipal waste generated. Control variables include environmental tax revenues as a share of GDP, log per capita GDP, log population density, and resource productivity in euros per kilogram. GDP per capita captures income-related waste generation and treatment capacity; population density captures settlement structure and collection conditions; environmental tax revenue proxies the broader fiscal-policy environment for environmental regulation; and resource productivity captures circular-economy performance beyond municipal recycling. The WFD compliance gap is defined as the positive difference between the legally binding recycling target and the country's actual rate, floored at zero for compliant countries.

China-specific National Sword exposure is constructed from UN Comtrade bilateral trade data: the annual volume in tonnes, converted from reported kilograms, of waste paper (HS 4707) and waste plastics (HS 3915) exported from each EU Member State to China over 2013–2017, averaged across the five pre-ban years. Luxembourg reported no bilateral exports to China under these codes and is assigned zero. Countries are classified as high exposure when their average pre-ban bilateral exports are strictly above the EU-27 median and as low exposure otherwise. Estonia equals the EU-27 median and is therefore included in the low-exposure group. As a robustness check, Luxembourg is excluded and the median is recomputed for the remaining 26 Member States; the classification of all non-Luxembourg countries remains unchanged (Table A4).

Landfill dependence is the share of municipal waste disposed to landfill in 2010, sourced from Eurostat and treated as a pre-determined structural variable. The ESIF absorption rate is total eligible expenditure divided by planned EU allocation under Thematic Objective 6 (Cohesion Fund and ERDF), from the European Commission ESIF 2014–2020 implementation data as reported for the 2023 programme year. Values may exceed 100% because final eligible expenditure can exceed initially planned allocations after programme reprogramming. Because the ESIF absorption indicator is measured once at the country level for the 2014–2020 programme, it is used only in the descriptive cross-country lock-in analysis. Table 1 presents descriptive statistics.

TABLE 1 | Descriptive statistics.

Variable	Mean	Std. Dev.	Min	Max
Municipal recycling rate (%)	32.6	17.4	0.5	71.4
WFD compliance gap (pp)	19.6	14.8	0.0	42.6
Environmental tax/GDP (%)	2.68	0.73	0.85	5.62
Resource productivity (EUR/kg)	1.69	1.06	0.28	5.81
Landfill dependency, 2010 (%)	53.8	26.4	0.4	94.3
ESIF TO6 absorption rate (%)	122.3	18.6	79.9	150.0
Pre-ban exports to China (kt/year, 2013–2017 average)	183.6	341.5	0.0	1375.3

3.2 | Empirical Strategy

This section sets out the empirical framework used to evaluate whether EU-27 municipal waste recycling rates converged over 2004–2023, whether that process stalled after 2018, and how catch-up speed covaried with policy pressure, external-shock exposure, and lock-in indicators. Consistent with the discussion in Section 3.1, let R_{it} denote the municipal waste recycling rate in country i and year t. The analysis combines sigma-convergence, absolute and conditional beta-convergence, the Phillips–Sul log-t test, formal structural break tests, exposure-based heterogeneity analysis, and lock-in mechanism evidence.

3.2.1 | Sigma-Convergence

Sigma-convergence assesses whether cross-country dispersion in recycling performance declines over time. Let the cross-sectional mean recycling rate in year t be defined as follows:

$$\bar{R}_t = \frac{1}{N_t} \sum_{i=1}^{N_t} R_{it} \tag{1}$$

The corresponding cross-sectional standard deviation is given by:

$$s_t = \left[\frac{1}{N_t - 1} \sum_{i=1}^{N_t} \left(R_{it} - \bar{R}_t \right)^2 \right]^{1/2} \tag{2}$$

Dispersion is measured using the coefficient of variation:

$$CV_t = \frac{s_t}{\bar{R}_t} \tag{3}$$

A decline in CV_t indicates sigma-convergence, whereas a constant or rising CV_t indicates stagnation or divergence. To estimate the average dispersion trend over a given period, the following time-series specification is used:

$$CV_t = \alpha_\sigma + \beta_\sigma t + u_t \tag{4}$$

A negative and statistically significant β_σ is interpreted as evidence of sigma-convergence. To examine whether convergence stalled after a candidate break year τ, the paper also estimates a segmented trend specification:

$$CV_t = \alpha + \beta_1 t + \beta_2 D_t^{(\tau)} + \beta_3 \left[(t - \tau) D_t^{(\tau)} \right] + \varepsilon_t \tag{5}$$

where $D_t^{(\tau)} = 1$ for $t \geq \tau$ and 0 otherwise. In this setup, β_1 is the pre-break slope and $\beta_1 + \beta_3$ is the post-break slope. The key identification criterion is whether the post-break slope shifts from negative to approximately zero, rather than relying only on the maximum Chow F-statistic.

3.2.2 | Absolute and Conditional Beta-Convergence

1. Absolute beta-convergence

Absolute beta-convergence tests whether countries with lower initial recycling rates subsequently grow faster. Let R_{i0} and R_{iT} denote the initial and terminal recycling rates over a window of length T. The average annual log-growth rate is defined as:

$$\bar{g}_i = \frac{1}{T} \ln\left(\frac{R_{iT}}{R_{i0}} \right) \tag{6}$$

The cross-sectional beta-convergence regression is then:

$$\bar{g}_i = \alpha_\beta + \beta \ln\left(R_{i0} \right) + \varepsilon_i \tag{7}$$

A negative and statistically significant β indicates catch-up: countries starting from lower initial recycling rates grow faster on average. The implied speed of convergence can be recovered as:

$$\lambda = -\frac{1}{T} \ln(1 + T\hat{\beta}) \tag{8}$$

and the associated half-life is:

$$t_{1/2} = \frac{\ln 2}{\lambda} \tag{9}$$

2. Conditional beta-convergence

Absolute beta-convergence assumes a common steady state for all Member States. This is unlikely in the present context because recycling performance depends on heterogeneous policy, economic, and structural conditions. The annual growth rate is therefore modelled as the log difference:

$$g_{it} = \Delta ln\left(R_{it} \right) = \ln\left(R_{it} \right) - \ln\left(R_{i,t-1} \right) \tag{10}$$

The baseline two-way fixed-effects specification is:

$$g_{it} = \alpha_i + \tau_t + \beta \ln\left(R_{i,t-1} \right) + \mathbf{X}_{it}^{\mathrm{T}} \boldsymbol{\theta} + \varepsilon_{it} \tag{11}$$

where α_i are country fixed effects, τ_t are year fixed effects, and X_{it}' is a vector of controls. Country fixed effects absorb time-invariant country components, whereas year fixed effects absorb shocks common to all countries in a given year. Standard errors are clustered at the country level.

To assess whether the conditional catch-up relationship varies with regulatory pressure, the model includes the WFD compliance gap G_{it} and its interaction with the lagged recycling rate:

$$\begin{aligned} g_{it} = {} & \alpha_i + \tau_t + \beta \ln\left(R_{i,t-1} \right) + \gamma G_{it} \\ & + \delta_1 \left[\ln\left(R_{i,t-1} \right) \times G_{it} \right] + \mathbf{X}_{it}^{\mathrm{T}} \boldsymbol{\theta} + \varepsilon_{it} \end{aligned} \tag{12}$$

The marginal convergence coefficient is therefore:

$$\frac{\partial g_{it}}{\partial \ln\left(R_{i,t-1} \right)} = \beta + \delta_1 G_{it} \tag{13}$$

If $\beta < 0$ and $\delta_1 > 0$, convergence weakens as the compliance gap widens. This corresponds to the argument that moderate laggards catch up, but extreme laggards improve more slowly.

To assess whether the conditional convergence relationship varies with broader circular-economy readiness, resource productivity P_{it} and its interaction with the lagged recycling rate can be added:

$$\begin{aligned} g_{it} = {} & \alpha_i + \tau_t + \beta \ln\left(R_{i,t-1} \right) + \gamma G_{it} + \delta_1 \left[\ln\left(R_{i,t-1} \right) \times G_{it} \right] \\ & + \phi P_{it} + \delta_2 \left[\ln\left(R_{i,t-1} \right) \times P_{it} \right] + \mathbf{X}_{it}^{\mathrm{T}} \boldsymbol{\theta} + \varepsilon_{it} \end{aligned} \tag{14}$$

3.2.3 | Phillips–Sul Log-*t* Convergence Test

Sigma- and beta-convergence impose relatively restrictive notions of convergence. To allow for heterogeneous transition paths and possible club dynamics, the paper additionally applies the Phillips–Sul log-t test. The recycling rate is represented as:

$$R_{it} = \delta_{it} \mu_t \tag{15}$$

where μ_t is a common component and δ_{it} is a time-varying idiosyncratic loading. Convergence occurs if the factor loadings converge to a common value over time. The relative transition parameter is defined as:

$$h_{it} = \frac{R_{it}}{N^{-1} \sum_{i=1}^{N} R_{it}} \tag{16}$$

The cross-sectional variance of the transition parameters is then:

$$H_t = \frac{1}{N} \sum_{i=1}^{N} \left(h_{it} - 1 \right)^2 \tag{17}$$

If countries converge, H_t tends to zero as t increases. The Phillips–Sul log-t regression is:

$$\log\left(\frac{H_t}{H_1} \right) - 2 \log(\log t) = a + b \log t + u_t, \qquad t = \lfloor rT \rfloor, \lfloor rT \rfloor + 1, \ldots, T \tag{18}$$

Following the standard implementation, the regression is estimated on the last 70% of the sample, that is, $r = 0.3$, using heteroskedasticity- and autocorrelation-consistent standard errors. The null hypothesis is $b \geq 0$ against the one-sided alternative $b < 0$. At the 5% level, the null of convergence is rejected when the one-sided t-statistic for b is less than -1.65. The test is applied to the full period and to relevant subperiods in order to distinguish long-run convergence from post-shock disruption.

3.2.4 | Structural Break Tests and Exposure Heterogeneity

The year at which convergence stalled is identified using a formal structural break procedure applied to the sigma-convergence time series. For each candidate break year τ, a Chow test compares a restricted pooled model with an unrestricted model estimated separately before and after the candidate break. Let SSR_P denote the residual sum of squares from the pooled model and SSR_1 and SSR_2 those from the two subsample regressions. The Chow statistic is:

$$F(\tau) = \frac{\left[SSR_P - \left(SSR_1 + SSR_2\right)\right]/k}{\left(SSR_1 + SSR_2\right)/\left(n_1 + n_2 - 2k\right)} \tag{19}$$

where k is the number of estimated parameters in each subperiod regression and n_1 and n_2 are the corresponding sample sizes. A large $F(\tau)$ indicates parameter instability at candidate break year τ. However, the article does not select the break year solely by the maximum F-statistic. Instead, the economically relevant signal is the post-break dispersion slope from Equation (5):

$$\beta_\sigma^{\text{post}}(\tau) = \beta_1 + \beta_3 \tag{20}$$

The year of stalled convergence is identified as the candidate year at which the post-break slope shifts from negative to approximately zero.

To examine heterogeneous post-shock dynamics, the paper estimates group-specific post-2018 sigma trends for countries with high and low exposure to China's import ban. Let H_i be the high-exposure indicator defined from pre-ban bilateral exports to China. The post-2018 group-specific regressions are:

$$\mathrm{CV}_t^{\mathrm{H}} = \alpha_{\mathrm{H}} + \beta_{\mathrm{H}} t + u_t^{\mathrm{H}}, \quad t = 2018, \ldots, 2023 \tag{21}$$

$$\mathrm{CV}_t^{\mathrm{L}} = \alpha_{\mathrm{L}} + \beta_{\mathrm{L}} t + u_t^{\mathrm{L}}, \quad t = 2018, \ldots, 2023 \tag{22}$$

The estimates of β_H and β_L provide a descriptive comparison of post-2018 convergence trends across the two exposure groups; they do not constitute a formal test of the between-group difference.

3.2.5 | Lock-In Mechanism Evidence

The article interprets lock-in through two complementary channels. First, structural lock-in is proxied by pre-determined landfill dependence. The cross-sectional association between landfill dependence and long-run recycling growth is examined using:

$$\bar{g}_i = \alpha_{\mathrm{L}} + \rho_1 L_i^{(2010)} + v_i \tag{23}$$

where the average annual growth rate is again defined as in Equation (6). A negative ρ_1 indicates that countries with heavier historical reliance on landfill tend to record slower long-run recycling growth.

Second, institutional lock-in is proxied by ESIF absorption capacity. The co-location of structural and institutional barriers is described using:

$$A_i = \alpha_{\mathrm{A}} + \rho_2 L_i^{(2010)} + \omega_i \tag{24}$$

A negative coefficient indicates that countries with heavier landfill dependence also tend to absorb a smaller share of EU environmental infrastructure funding. These cross-sectional relationships do not identify causal effects. Rather, they document the co-location of pre-existing structural and institutional disadvantages across Member States.

4 | Main Results and Robustness

4.1 | Sigma-Convergence and the 2018 Structural Break

The CV declined from 0.810 in 2004 to 0.380 in 2023 (Table 2). The full-period linear trend was −0.028 per year ($p < 0.001$). Before 2018, the trend was −0.039 per year ($p < 0.001$). After 2018, the trend was −0.001 per year ($p = 0.782$), which is statistically indistinguishable from zero. The Chow test for a break at 2018 yielded $F = 35.7$ ($p < 0.001$).

Panel A of Figure 1 presents the placebo year scan. The Chow F-statistic peaks in 2016 at $F = 44.8$. However, this outcome does not identify 2016 as the break year. On a monotone time series, break tests yield higher F-statistics for candidate years closer to the sample centre because the contrast between the pre- and post-period regression lines is maximised. The relevant identification signal is the post-break slope coefficient. The slope is negative for all candidate years from 2010 to 2017, crosses to near zero at 2018, and turns positive from 2019 onward. The year 2018 is therefore identified as the year in which convergence stalled.

Panel B of Figure 1 presents the bilateral trade exposure to China analysis. Using the strict above-median split, the high-exposure group comprises 13 Member States and the low-exposure group 14 Member States. After 2018, the sigma slope was −0.0036 per year for the high-exposure group (SE = 0.0022, $p = 0.171$) and +0.0022 per year for the low-exposure group (SE = 0.0029, $p = 0.493$). Neither post-2018 trend is statistically distinguishable from zero.

TABLE 2 | Sigma-convergence results.

Period/Group	CV start	CV end	Sigma trend (beta, /yr)
Full sample, 2004–2023	0.810	0.380	−0.028*** (0.002)
Pre-ban, 2004–2017	0.810	0.391	−0.039*** (0.004)
Post-ban, 2018–2023: all countries	0.391	0.380	−0.001 (0.005)
Post-ban: High exposure to China ($n = 13$)	0.345	0.329	−0.0036 (0.0022)
Post-ban: Low exposure to China ($n = 14$)	0.437	0.434	+0.0022 (0.0029)

Note: OLS on annual CV series; standard errors are reported in parentheses. Exposure is average 2013–2017 bilateral exports to China under HS 4707 and 3915 (UN Comtrade). Countries strictly above the EU-27 median are classified as high exposure ($n = 13$); Estonia equals the median and is classified as low exposure ($n = 14$). High-exposure countries are Belgium, Croatia, Czechia, France, Germany, Greece, Ireland, Italy, the Netherlands, Poland, Portugal, Slovenia, and Spain. Chow test for a break at 2018: $F = 35.7$ ($p < 0.001$). ***$p < 0.01$, **$p < 0.05$, *$p < 0.10$.

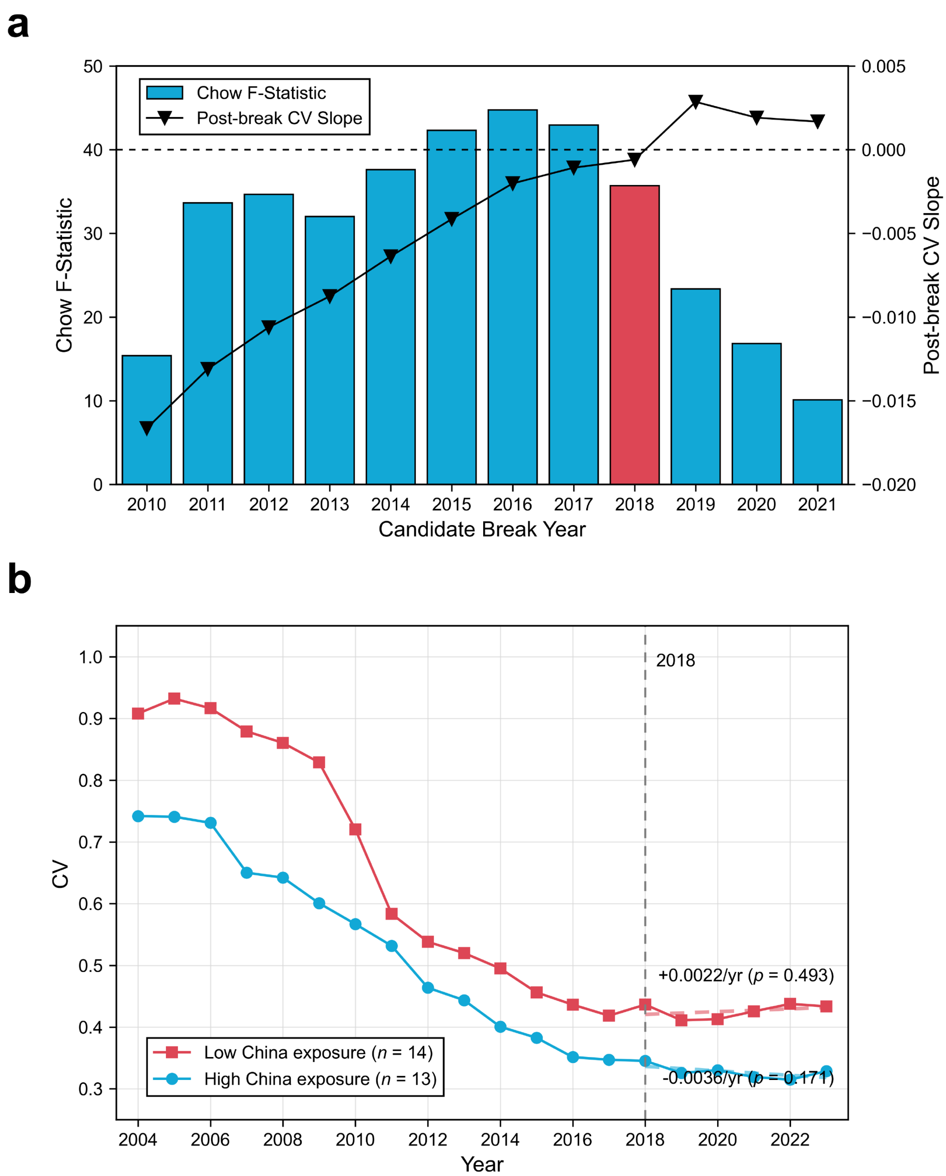

FIGURE 1 | Structural break evidence. Panel A shows Chow F-statistics (bars, left axis; the red bar marks 2018) and post-break CV slope coefficients (triangles, right axis) for candidate break years 2010–2021. Panel B shows the annual CV paths for high-exposure countries ($n = 13$) and low-exposure countries ($n = 14$), classified using average 2013–2017 bilateral exports to China under HS 4707 and 3915 and a strict above-median rule. Dashed lines show post-2018 OLS trends.

4.2 | Beta-Convergence

Table 3 presents the beta-convergence results. The absolute beta-convergence coefficient is −0.040 (SE = 0.004, $p < 0.001$), with $R^2 = 0.833$. The implied convergence speed is $\lambda = 0.076$ per year, corresponding to a half-life of 9.1 years.

Panel B of Figure 2 shows the cross-sectional scatter of 2004–2023 log-growth rates against 2004 initial log-values. Lithuania (+47.4 pp), Latvia (+46.4 pp), and Slovakia (+44.2 pp) show the largest absolute gains, all starting below 10% in 2004. Conditional beta-convergence in a two-way fixed-effects (TWFE) specification without controls yields $\beta = -0.157$ ($p < 0.001$), corresponding to a half-life of 4.1 years. Adding structural controls yields $\beta = -0.195$ ($p < 0.001$) and a conditional half-life of 3.2 years.

TABLE 3 | Beta-convergence results.

Specification	Beta	Lambda (/yr)	Half-life (years)
Absolute (cross-section, $N = 27$)	−0.040*** (0.004)	0.076	9.1
Conditional TWFE, no controls	−0.157*** (0.030)	0.171	4.1
Conditional TWFE, with controls	−0.195*** (0.048)	0.216	3.2

Note: Absolute: cross-sectional OLS, $R^2 = 0.833$. Conditional: two-way fixed-effects panel OLS, $N = 513$, standard errors clustered by country. ***$p < 0.01$.

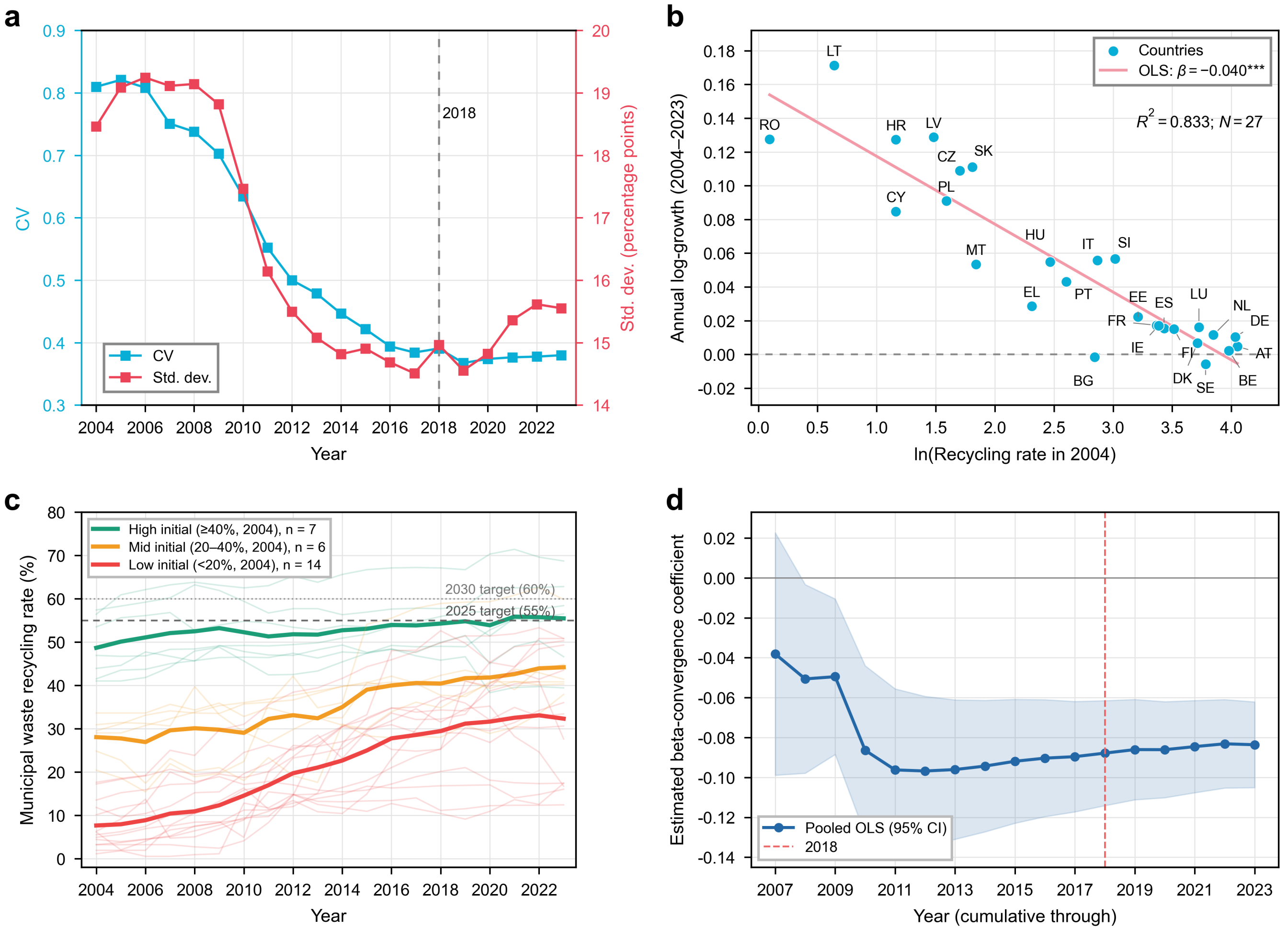

FIGURE 2 | EU-27 municipal waste recycling rate convergence, 2004–2023. Panel A shows the cross-sectional coefficient of variation (left axis) and standard deviation (right axis) over 2004–2023. Panel B shows the absolute beta-convergence scatter ($N = 27$, $\beta = -0.040$***, $R^2 = 0.833$). Panel C shows country-level trajectories grouped by 2004 initial level; thick lines are group means; dashed horizontal lines mark the 2025 (55%) and 2030 (60%) WFD targets. Panel D shows the beta-convergence coefficient estimated over expanding cumulative windows; the vertical line marks 2018.

We also test whether the absolute beta-convergence result is sensitive to spatial dependence. Using two row-standardised inverse-distance matrices based on geographic distance and initial GDP-per-capita distance, respectively, we estimate spatial lag and spatial error models. The beta-convergence coefficient remains stable across all spatial specifications, ranging from −0.0399 to −0.0418, compared with −0.0402 in the baseline model. Spatial lag effects are insignificant under both matrices; the spatial error term is significant only under the geographic matrix, indicating geographically clustered residual shocks rather than a change in the convergence estimate. These results, reported in Table A5, show that the absolute beta-convergence estimate is stable across the reported spatial specifications.

4.3 | Phillips–Sul Log-*t* Test

Table 4 presents the Phillips–Sul results. The full-period test (2004–2023) returns $b = 0.152$ and $t = 1.32$: the null hypothesis of full-panel convergence is not rejected. The pre-ban period (2004–2017) returns $b = 0.420$ and $t = 7.02$, confirming strong convergence. The post-ban period (2018–2023) returns

TABLE 4 | Phillips–Sul log-*t* convergence test results.

Period	*b*-hat	*t*-statistic	Conclusion
Full period, 2004–2023	0.152	1.322	Do not reject convergence
Pre-ban, 2004–2017	0.420	7.023	Strong convergence
Post-ban, 2018–2023	−1.780	−17.578	Reject convergence
Post-ban, 2018–2023, excl. 2020	−1.917	−20.415	Reject convergence (robust)

Note: Heteroskedasticity- and autocorrelation-consistent standard errors, trim factor $r = 0.3$. Critical value at the 5% significance level: $t = -1.65$ (one-sided). DNR = do not reject. Club clustering identifies one club covering all 27 Member States.

$b = -1.780$ and $t = -17.6$, decisively rejecting convergence. Excluding 2020 to control for the WFD accounting methodology change and COVID disruption yields $b = -1.917$ and $t = -20.4$. The full-period result masks a strong convergence phase that was disrupted by the 2018 structural break. The club clustering algorithm identifies a single convergent club covering all 27 Member States: no permanent divergent subgroup has formed during the sample period.

The analysis covers the EU-27 Member States over 2004–2023. The United Kingdom is excluded from the main panel because it left the EU in 2020 and is no longer subject to the Waste Framework Directive target system examined here; moreover, Eurostat does not provide a consistent UK municipal recycling series for the full sample period. As a sensitivity check, we add the UK using Defra's 'Waste from Households' recycling rate for the overlapping 2010–2022 period. The sigma- and beta-convergence results are materially unchanged, and the results are reported in Table A3.

Because the log-t result can be sensitive to the trim parameter, we report sensitivity to $r = 0.20$, 0.25, 0.30, and 0.33 in Table A1. The post-ban rejection of convergence is invariant across all four trim rates ($b = -1.780$, $t = -17.58$ throughout). The full-period and pre-ban conclusions are also robust, except that at $r = 0.20$ the pre-ban statistic loses precision because only a subset of the 14 pre-ban observations enters the regression, a known small-sample property of the test. We retain $r = 0.30$ as the baseline.

4.4 | Policy Correlates and Convergence Barriers

Table 5 presents the conditional beta-convergence regressions. M1 reports the baseline conditional catch-up estimate. M2 adds structural controls. M3 introduces the WFD compliance gap. M4 adds the interaction between the lagged recycling rate and the compliance gap. M5 adds resource productivity and its interaction.

Adding the WFD compliance gap raises $R2$ from 0.199 in M2 to 0.374 in M3. The full non-linear specification in M4 reaches $R^2 = 0.759$. The positive interaction coefficient ($\delta_1 = +0.028$, $p < 0.001$) implies that the conditional convergence coefficient becomes less negative as the compliance gap increases. Panel A of Figure 3 illustrates this: countries near the target show the steepest convergence slope, while countries far from the target show a much flatter relationship between initial level and growth. This pattern is consistent with the interpretation that structural barriers weaken the association between regulatory pressure and recycling growth.

Resource productivity in M5 enters with a positive but imprecisely estimated coefficient (+0.162, $p = 0.114$). The interaction with the lagged recycling rate is negative but not significant ($p = 0.219$). The estimates therefore provide inconclusive evidence that resource productivity modifies convergence speed; the level coefficient is positive but imprecisely estimated.

To ensure the policy-driver coefficients are not distorted by the 2020 pandemic year or the accompanying reporting-methodology change, we re-estimated all panel specifications excluding 2020 (Table A2). The coefficients are virtually unchanged: in M4 the beta-convergence coefficient moves from −1.831 to −1.859 and the WFD non-linearity term from +0.028 to +0.029, while the sample falls from 513 to 486 observations. Thus, excluding 2020 leaves the signs and magnitudes of the key coefficients largely unchanged.

Figure 4 presents the cross-sectional mechanism evidence. Panel A shows that countries with higher 2010 landfill dependence have lower recycling rates in 2023 ($r = -0.63$, $p < 0.001$). Panel B shows that those same countries absorb less of their EU environmental infrastructure funding ($r = -0.78$, $p < 0.001$). Together, these two panels characterise the convergence barriers in the bottom quartile of Member States: high structural lock-in (entrenched landfill infrastructure) and weak institutional implementation capacity appear to be co-located and jointly limit the translation of regulatory pressure and financial transfers into recycling improvement.

As an additional indicator of landfill-related policy constraints, we examine landfill taxes on municipal waste using the European Environment Agency overview, cross-checked against the OECD Policy Instruments for the Environment database. The EEA source reports a positive landfill tax for 17 of the 26 Member States it covers; nine countries have no landfill tax, and Croatia is not listed. Because this indicator is effectively time-invariant at the country level over the study period, it cannot be separately identified in the two-way fixed-effects specification with country fixed effects and is therefore treated as supplementary cross-sectional evidence.

TABLE 5 | Conditional beta-convergence: Policy correlates.

Dependent variable	M1 Baseline	M2 +Controls	M3 +WFD	M4 +WFD × lag	M5 +Res. prod.
β: $\ln(R_{\{i,t\text{-}1\}})$	−0.157*** (0.030)	−0.195*** (0.048)	−0.405*** (0.069)	−1.831*** (0.244)	−1.788*** (0.244)
γ: WFD compliance gap	—	—	−0.021*** (0.003)	−0.136*** (0.019)	−0.135*** (0.019)
δ_1: lag × WFD gap	—	—	—	+0.028*** (0.005)	+0.028*** (0.005)
Resource productivity	—	—	—	—	+0.162 (0.103)
δ_2: lag × Res. productivity	—	—	—	—	−0.035 (0.029)
N	513	513	513	513	513
R^2	0.183	0.199	0.374	0.759	0.761

Note: Two-way fixed-effects panel OLS with country and year fixed effects. Standard errors clustered by country are reported in parentheses. Controls in M2–M5 (ln GDP per capita, ln population density, and environmental tax/GDP) are included but not reported for brevity. * $p < 0.10$, ** $p < 0.05$, *** $p < 0.01$.

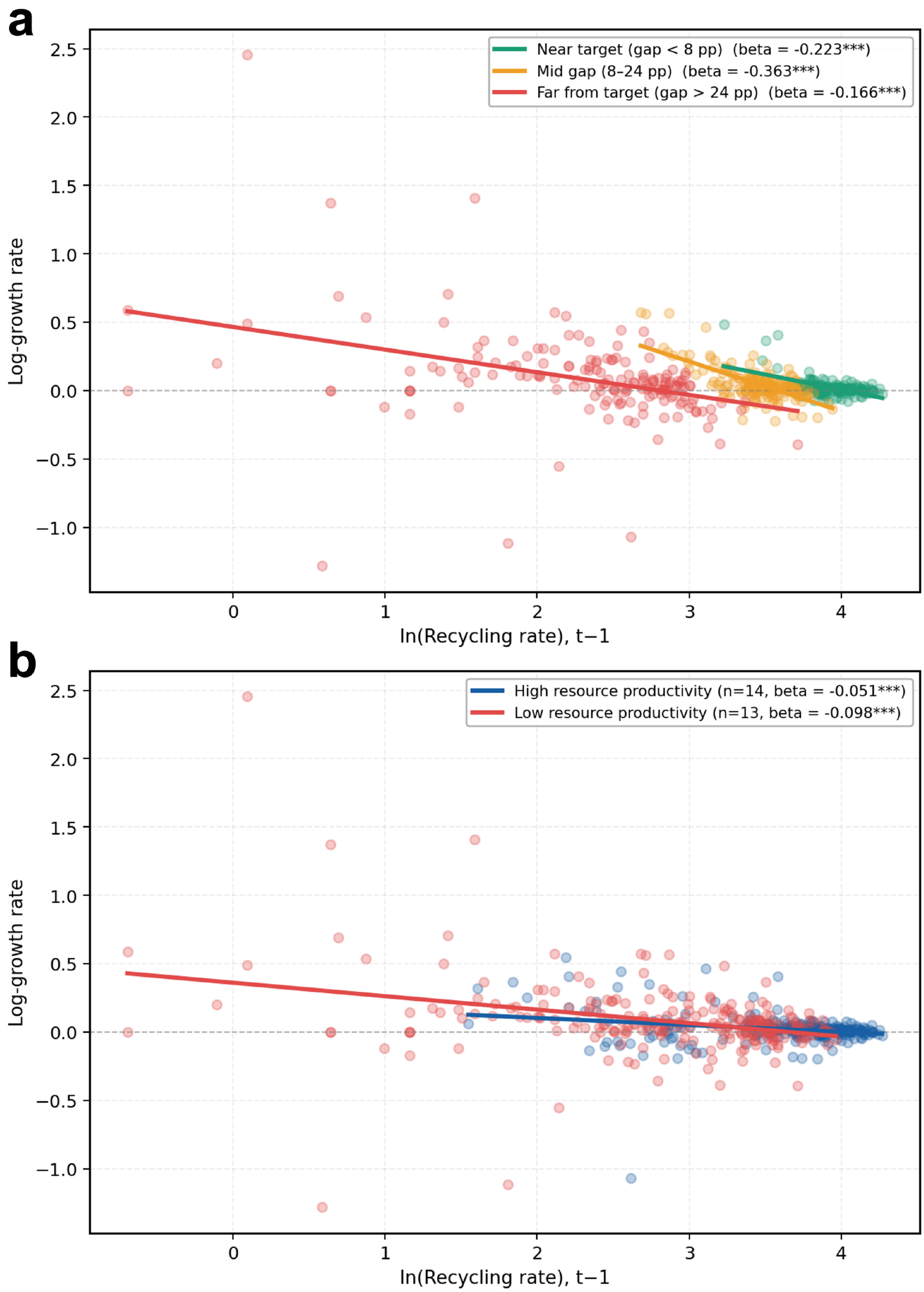

FIGURE 3 | Policy correlates of convergence. Panel A shows OLS convergence slopes for groups defined by distance to WFD target (near: Gap < 8 pp.; mid: 8–24 pp.; far: Gap > 24 pp). Panel B shows convergence slopes by resource productivity group (split at EU-27 median).

The pattern is informative but not statistically decisive. In 2023, countries with a landfill tax had a higher average recycling rate than those without one, 44.0% versus 35.7%, but the difference is not statistically significant ($t = 1.09$, $p = 0.30$). Among taxing countries, the landfill-tax rate is positively but not significantly correlated with the 2023 recycling rate ($r = 0.44$, $p = 0.08$). This comparison is consistent with landfill taxes being one possible route to landfill diversion, but it does not establish that such taxes are necessary for high recycling performance: Germany, the highest recycler in the sample, relies on a direct landfill ban for untreated municipal waste rather than a landfill tax. The evidence therefore complements the landfill-dependency results by showing that convergence barriers are better captured by entrenched disposal structures and implementation capacity than by the presence of a single fiscal instrument.

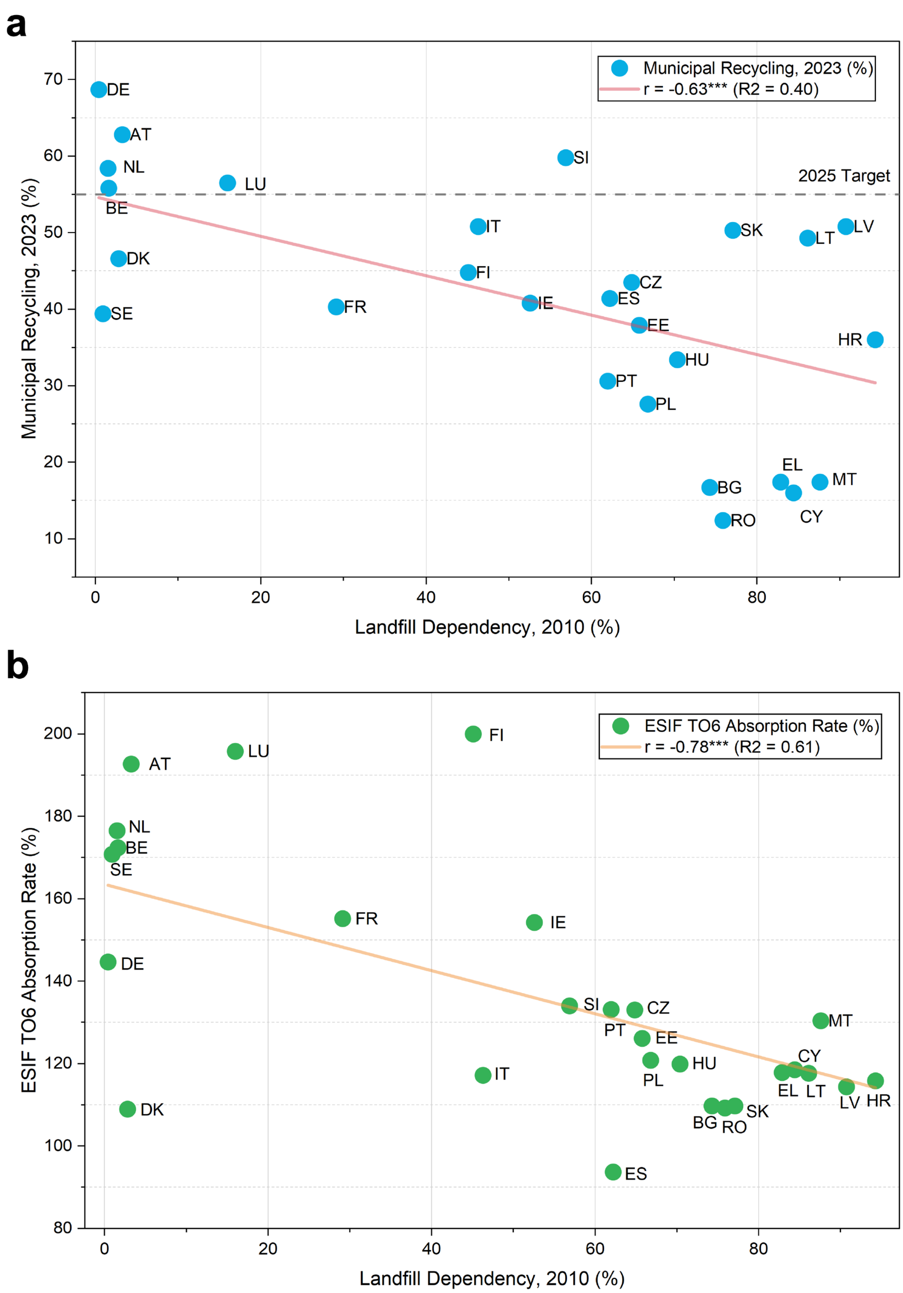

FIGURE 4 | Structural and institutional barriers to convergence. Panel A shows the cross-sectional scatter of 2023 recycling rates against 2010 landfill dependence ($r = -0.63$, $p < 0.001$). Countries with higher historical landfill dependence are associated with lower current recycling rates. Panel B shows ESIF Thematic Objective 6 absorption rates against 2010 landfill dependence ($r = -0.78$, $p < 0.001$). Countries with higher landfill dependence absorb less of their EU environmental infrastructure allocation. Both panels are country-level cross-sections; lines show OLS fits.

5 | Discussion

5.1 | The Convergence Process and Its Disruption in 2018

EU-27 recycling rates converged substantially over 2004–2023. The CV declined by 53%; the absolute catch-up regression explains 83% of cross-country growth variation, and the Phillips–Sul test does not reject full-panel convergence over the full period. The conditional estimates show a strong association between WFD compliance pressure and catch-up across Member States.

The analysis identifies a structural break in 2018. The Chow test confirms a significant change in the sigma-convergence trend ($F = 35.7$, $p < 0.001$). The placebo year scan clarifies the identification logic: the relevant signal is the post-break slope, not the F-statistic maximum. The F-statistic peaks in 2016 because candidate break years near the sample centre yield the largest contrasts in a monotone series. The post-break slope becomes nearly zero in 2018, which is the year in which convergence stalled.

The bilateral trade exposure analysis provides descriptive country-level evidence consistent with a disruption in convergence after 2018. Under the strict above-median specification, the post-2018 sigma slopes are −0.0036 per year for the high-exposure group and +0.0022 per year for the low-exposure group, and neither is statistically significant. The results therefore indicate stalled convergence in both exposure groups; they do not show that one group experienced a statistically significant greater change than the other.

5.2 | Non-Linear Catch-Up and Convergence Barriers

The WFD compliance gap is the strongest reported correlate of conditional convergence, but the positive interaction term ($\delta_1 = +0.028$, $p < 0.001$) implies a less negative conditional convergence coefficient for countries farther from the target. Bulgaria, Romania, and Cyprus were 38–43 percentage points below the 2025 target as of 2023. The marginal beta-convergence coefficient at these compliance gap levels becomes less negative. The estimates do not indicate rapid catch-up at the largest compliance gaps, where structural barriers are also most pronounced.

The cross-sectional patterns in Figure 4 jointly characterise two dimensions of convergence barriers. Panel A shows that countries with higher 2010 landfill dependence have lower 2023 recycling rates ($r = -0.63$): the lock-in from entrenched landfill infrastructure is visible in the persistence of low recycling levels. Panel B shows that these same countries also absorb less of their EU environmental infrastructure funding ($r = -0.78$): the institutional capacity to deploy corrective investments is weakest precisely where structural transformation needs are largest. These two patterns are descriptive cross-sectional associations, not causal estimates. They jointly characterise the barriers facing the bottom quartile of Member States. Two mechanisms may underlie this co-location. Entrenched landfill infrastructure may reduce incentives to shift toward recycling-related investment because sunk costs, existing contracts, and local disposal routines create path dependence. At the same time, weaker administrative capacity may simultaneously contribute to both lower ESIF absorption and slower waste-system transformation. The present analysis cannot separate these channels causally, but the observed co-location is consistent with mutually reinforcing constraints in lagging Member States.

5.3 | Comparison With Previous Literature

The results are broadly consistent with previous studies on EU waste-management performance, but they extend this literature in three ways. First, the finding of overall convergence in EU-27 municipal waste recycling rates is consistent with (Marin et al. 2018), who document catching-up in EU waste management. At the same time, this convergence does not imply full homogenisation of waste-management systems. The persistent differences emphasised by Castillo-Giménez et al. (2019) and the regional target gaps identified by Sastre et al. (2018) remain relevant. The pattern observed in this study is therefore best interpreted as 'convergence with continuing heterogeneity': lagging Member States generally improved faster, but national and sub-national differences in infrastructure, policy instruments, and implementation capacity did not disappear.

Second, this study extends the literature on China's National Sword policy by linking external trade disruption to the internal convergence path of EU recycling performance. Brooks et al. (2018), Wen et al. (2021), and Tran et al. (2021) show that China's import restrictions reshaped global waste trade and altered the environmental consequences of recyclable-material flows. The present results show that this shock was not only a trade-flow event, but also a convergence disturbance within the EU. The 2018 structural break and the stalled post-2018 convergence in both exposure groups indicate that dependence on external outlets for recovered paper and waste plastics affected Member States' relative catch-up dynamics. This adds a convergence-based interpretation to the existing trade and environmental-impact literature.

Third, the nonlinear association between WFD target pressure and catch-up helps clarify the limitations of uniform targets without rejecting their policy relevance. The results show that target pressure is associated with faster catch-up overall, but that this association weakens for countries farthest from the target. This finding is consistent with Salmenperä et al. (2021)), who argue that circular-economy progress depends on institutional, market, technological, and behavioural conditions, and with Albizzati et al. (2024), D'Adamo et al. (2024), and (Hondroyiannis et al. (2024), who show that municipal waste performance is shaped by system design and socioeconomic context. The translation of EU targets into recycling improvements depends on the capacity of national and local systems to respond.

Finally, the evidence on landfill dependence and ESIF absorption capacity provides an empirical expression of the lock-in mechanisms discussed in the waste-management literature. Wilson (2023), Campitelli et al. (2022), and Ntostoglou et al. (2025) emphasise that waste systems are shaped by historically embedded infrastructures, routines, and institutional arrangements. Incaltarau et al. (2020) and Dotti et al. (2024) further show that administrative capacity affects EU fund absorption and policy implementation. The present study connects these two strands by showing that structural lock-in and institutional implementation capacity are co-located in the recycling-convergence process. However, this should be interpreted with caution as evidence of associated barriers rather than as a fully identified causal sequence. Overall, the contribution of this paper is not simply to show that EU waste performance differs across countries, but to document when convergence occurs, or stalls and which observed constraints are associated with slower catch-up.

5.4 | Policy Implications

Three policy considerations follow from the analysis. First, the 2018 disruption is now formally established. The exposure patterns support considering measures that strengthen domestic secondary-material markets and reduce reliance on external outlets, although the analysis does not estimate the causal effect of such measures. Second, convergence is non-linear with respect to compliance distance. A uniform 2025 deadline is not well matched to the structural position of approximately 20 Member States still below 55%. The estimates support considering differentiated compliance pathways linked to infrastructure and administrative capacity milestones, subject to feasibility, distributional, and legal constraints beyond this analysis. Third, structural barriers and institutional implementation capacity are co-located: the countries that most need waste infrastructure investment are also the least able to absorb the EU funds allocated for that purpose. This co-location supports consideration of coordinated financial transfers, technical assistance, and administrative capacity building, although the analysis does not compare the effectiveness or cost of these instruments.

5.5 | Limitations

Three limitations apply, first, the bilateral exposure-to-China measure uses HS codes 4707 and 3915 at the 4-digit level. More granular 8-digit subcodes would allow differentiation by material quality and purity. Second, the ESIF evidence is based on a country-level cross-sectional association and does not identify the effect of fund absorption on recycling growth. Third, the analysis is at the national level, which cannot capture sub-national heterogeneity in recycling infrastructure documented in the circular economy literature.

6 | Conclusion

EU-27 municipal waste recycling rates converged strongly over 2004–2023, driven by WFD target pressure. This convergence stalled in 2018. As of 2023, only 6 of 27 Member States had achieved the 2025 target of 55%.

Three conclusions follow. First, the Chow test confirms a structural break in 2018 ($F=35.7$, $p<0.001$), and the placebo-year scan shows that the post-break sigma slope approaches zero in that year. The exposure-group analysis yields statistically insignificant post-2018 trends for both high-exposure countries (−0.0036 per year, $p=0.171$) and low-exposure countries (+0.0022 per year, $p=0.493$), consistent with a broad stalling of convergence.

Second, convergence is non-linear with respect to compliance distance. The positive interaction between the initial level and compliance gap ($\delta_1=+0.028$, $p<0.001$) implies weaker conditional convergence at larger compliance gaps. The results suggest that differentiated compliance pathways may be more consistent with the observed convergence dynamics than a uniform 2025 deadline.

Third, structural and institutional barriers to convergence are co-located. Countries with the highest landfill dependence also have the lowest ESIF absorption rates ($r=-0.78$). This descriptive cross-country relationship supports considering both waste infrastructure and institutional implementation capacity when designing convergence support.

Author Contributions

Lianghan Cong: conceptualization, methodology, formal analysis, investigation, data curation, writing – original draft. **Lan Liu:** methodology, validation, writing – review and editing. **Shuaiyi Lu:** data curation, formal analysis, writing – review and editing. **Pan Jiang:** validation, writing – review and editing. **Haneda Katsuyuki:** writing – review and editing. **Xiaoshu Lü:** supervision, project administration, writing – review and editing.

Acknowledgements

The authors have nothing to report. Open access publishing facilitated by Vaasan yliopisto, as part of the Wiley - FinELib agreement.

Funding

This work was supported by the Research Council of Finland, DISCAS Project (Grant No. 362751), Sustain School Project (Grant No. 359189).

Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data used in this study are derived from publicly available Eurostat datasets.

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Appendix A

TABLE A1 | Phillips–Sul log-t sensitivity to trim factor r.

Period	$r = 0.20$	$r = 0.25$	$r = 0.30$	$r = 0.33$
Full 2004–2023	+0.208/+2.63	+0.218/+2.05	+0.152/+1.32	+0.152/+1.32
Pre-ban 2004–2017	−0.029/−0.18	+0.206/+1.60	+0.420/+7.02	+0.420/+7.02
Post-ban 2018–2023	−1.780/−17.58	−1.780/−17.58	−1.780/−17.58	−1.780/−17.58
Post-ban excl. 2020	−1.917/−20.42	−1.917/−20.42	−1.917/−20.42	−1.917/−20.42

Note: Each cell reports b-hat/t-statistic. Do not reject convergence if $t \geq -1.65$. Post-ban results are invariant to the trim rate. At $r = 0.20$ the pre-ban period ($T = 14$) loses precision.

TABLE A2 | Robustness: Panel regressions excluding 2020.

Specification	Sample	beta	delta_1	N	$R2$
M4: +WFD × lag	incl. 2020	−1.831***	+0.028***	513	0.759
M4: +WFD × lag	excl. 2020	−1.859***	+0.029***	486	0.763
M5: +Res. prod.	incl. 2020	−1.788***	+0.028***	513	0.761
M5: +Res. prod.	excl. 2020	−1.815***	+0.028***	486	0.766

Note: TWFE with country and year fixed effects, standard errors clustered by country; all controls included but not shown. Coefficients are stable and marginally strengthened when 2020 is excluded.

TABLE A3 | Robustness: UK sensitivity using Defra Waste from Households data (2010–2022).

Test (2010–2022 window)	EU-27 only	EU-27 + UK	Difference
Sigma-convergence slope (/yr)	−0.0187***	−0.0182***	0.0005
Beta-convergence coefficient	−0.0584***	−0.0586***	0.0002
Number of countries	27	28	+1 (UK)

Note: UK data are the Defra 'Waste from Households' recycling rate (Official Statistics of UK 2025), the only official UK series available across this window. This measure is defined more narrowly than the Eurostat municipal waste indicator used in the main panel, so the check confirms robustness to UK inclusion rather than extending the main sample. The UK series covers 2010–2022 only; the comparison is therefore restricted to that window. The UK rate rises from 40.4% in 2010 to 43.1% in 2022. ***$p < 0.01$.

TABLE A4 | Robustness: Exposure to China group assignment and Luxembourg.

Setting	Group	N	Post-2018 sigma slope (p)
A: Luxembourg assigned zero (27)	High exposure	13	−0.0036 (0.17)
A: Luxembourg assigned zero (27)	Low exposure	14	+0.0022 (0.49)
B: Luxembourg removed (26)	High exposure	13	−0.0036 (0.17)
B: Luxembourg removed (26)	Low exposure	13	+0.0010 (0.77)

Note: Exposure to China is average bilateral exports to China under HS 4707 and 3915 over 2013–2017 (UN Comtrade). Luxembourg is the only Member State with no bilateral export record. In Setting A it is assigned zero (low-exposure group); in Setting B it is removed and the median recomputed on 26 countries. High/low membership of all 26 non-Luxembourg countries is identical across settings. The high-exposure post-2018 slope is unchanged; the low-exposure slope is positive, near zero, and not significant in either setting. The core result is unaffected.

TABLE A5 | Robustness: Spatial models of absolute beta-convergence.

Model	beta (initial ln)	Spatial parameter	Sig.
OLS (baseline)	−0.0402***	—	—
SAR, geographic W	−0.0403***	$\rho = -0.088$	$p = 0.79$
SAR, economic W	−0.0399***	$\rho = +0.026$	$p = 0.85$
SEM, geographic W	−0.0418***	$\lambda = +0.714$	$p < 0.001$
SEM, economic W	−0.0411***	$\lambda = +0.348$	$p = 0.12$

Note: Dependent variable is the annualised 2004–2023 growth in the log recycling rate; regressor is the initial (2004) log recycling rate; $N = 27$. Geographic weights are row-standardised inverse great-circle distances between national capitals; economic weights are row-standardised inverse absolute differences in initial (2004) log per capita GDP. SAR = spatial lag model; SEM = spatial error model, estimated by maximum likelihood. Moran's I on OLS residuals: 0.11 ($p = 0.06$) geographic, 0.14 ($p = 0.19$) economic. The beta coefficient is stable across all specifications (maximum deviation under 4%); spatial lag effects are insignificant. ***$p < 0.01$.