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Regionalized life-cycle monetization method to thrive sustainable transition of rural food waste management in China

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Abstract

Monetization of externalities has served to thrive sustainable technology transition towards a low carbon scoiety. However, regional monetization method is lacking in China. Herein, Chinatax^{RCP} was built to enable an environmental-economic (E-E) integration and analysis of environmental externalities in different medium (air-water-soil-human), facilitating decision making after considering tradeoffs among environmental impacts or between environment and economics. As a case study, Chinatax^{RCP} was applied on evaluating technology-transition risks from landfill for rural FW management in Zhejiang who is a pioneer practicing "zero-waste city" in China. E-E integration results using Chinatax^{RCP} indicated that biodrying & maturity, and two biotransformation options were promising. However, after transition from landfill, environmental pollution in air-soil-water and resource-related indicators were mostly relieved but with compromise on human health, which indicates efforts are required to control potential risks. Furthermore, policy scenarios revealed that subsidies are crucial for FW management and extra leverage should be reinforced to decouple them.

Abbreviation

Items	Abbr.	Items	Abbr.
Food waste	FW	Human carcinogenic toxicity	HT_c
Anaerobic digestion	AD	Ozone formation, Terrestrial ecosystem	EOFP
Greenhouse gas	GHG	Stratos. ozone depletion	ODP

Life cycle assessment	LCA	Fossil resource scarcity	FFP
Monetization factor	MF	Mineral resource scarcity	SOP
Black soldier fly	BSF	Land use	LU
Red head fly	RHF	Ozone formation, Human health	HOFP
Environmental-economic	E-E	Net present value	NPV
Environmental Prices	EP	Business as usual (landfill)	BAU
Global warming potential	GWP	Functional unit	FU
Fine particulate matter formation	PMFP	Environmental impact monetization	EIM
Terrestrial acidification	TAP	Marine ecotoxicity	METP
Water consumption	WCP	Human non-carcinogenic toxicity	HT_{nc}
Freshwater eutrophication	FEP	Ionizing radiation	IR
Marine eutrophication	MEP	Ozone Depleting Substances	ODS
Freshwater ecotoxicity	FETP	Life-cycle costing	LCC
Terrestrial ecotoxicity	TETP	Supplementary Information	SI
		Supplementary Dataset	SD

Main

Handling food waste (FW) from a circular economy perspective is one of the major strategies to achieve a sustainable food system to reduce environmental impacts, improve resources efficiency from FW disposal, and to meet the nutritional demand of a growing populations [1, 2]. In China, 27% of food for human consumption (349 ± 4 million tonne) is lost annually [3], among which approximately 140–196 Mt is collected as FW [4]. Around one third of FW is in rural areas [5-7], making it difficult to be collected, transported, and centralized for large-scale treatment i.e. anaerobic digestion (AD, > 100 t/d) [4, 8, 9]. Currently, rural FW is disposed of via landfill disposal and incineration which emit approximately 38 Mt CO_2 eq. in total, accounting for 6% of greenhouse gas (GHG) emissions of the Chinese agriculture system [3]. Mismanagement of rural FW would also pose risks to the environment and human health via other pollution [10]. On the other hand, rural FW contains protein, oil, and nutrients (NPK), offering potential to be reused as resources by small-scale AD, composting, protein production, and pyrolysis for biochar, etc. However, technology transitions from conventional waste disposal to reutilization may accompany tradeoffs among different environmental issues, as well as among environmental, economic, and/or other issues, bringing about inequities/environmental risks by transmedia/transregional migration of pollution [11-14].

To ensure the sustainable transition of technologies, environmental externalities quantified via monetization could be one of the economic indicators to reveal the above-mentioned tradeoffs and inequities, and facilitate decision-making to manage environmental risks of potential internalization and to reduce externalities voluntarily [15]. Currently, externalities' monetization methodologies have been developed in Europe and Japan, i.e., Ecovalue, Ecotax, True Price,

LIME, Environmental Prices, etc [16-20]. They have been used and recognized as particularly relevant for the food sector whose dependency on both nature and people is direct [18]. However, a systematic framework in China is lacking though efforts have been made [18, 21, 22].

Herein, following monetization standard ISO14008-2019, we proposed a Chinese regionalized life-cycle monetization method (named as Chinatax^{RCP}), which is applicable for ReCiPe 2016 indicators in life cycle assessment (LCA) [23]. Monetization factors (MF) for each environmental indicator were compared with other existing monetization methods after normalization. Furthermore, Chinatax^{RCP} was applied in evaluating the transition of rural FW management from landfill disposal to reuse in Zhejiang province, a leader practicing the 'zero waste city' policy in China, as a case study. Ten FW-to-resource technologies evaluated were investigated. T1: mechanical dehydration, T2: biodrying and maturity, and T3: solar-assisted composting could convert rural FW into compost [24]. T4: underground AD is tailored for rural areas with a small capacity (1–10 t/d) to produce biogas. T5: heat pyrolysis carbonization could cotreat FW with agricultural waste to produce biochar. Biotransformation for black soldier fly (BSF) larvae or Redhead fly (RHF) (T6–T9) upcycles FW into animal feed, compost, and/or biooil. T10: enzyme production. These ten demonstrated rural FW-to-resource technologies were compared via LCA and an Environmental-Economic (E-E) integration analysis by considering externalities as a economic factor using Chinatax^{RCP}. Finally, a transmedia migration analysis was proposed to investigate the potential tradeoffs among different groups of interested media affected by technological transition to support decision making and environmental risk analysis. In addition, scenario analysis of product substitution and policy changes was performed to support the robustness of conclusions.

Results

Establishment of regionalized life-cycle monetization method in China. The regionalized life-cycle monetization model in 31 provinces in China, named Chinatax^{RCP}, was built following ISO14008-2019 guidelines [23] on externalities' monetization (Fig.1), and compared with other nine worldwide-existing methods (Fig.2). As shown in Fig. 2(a), approximately ten monetization methods which are applied to different countries based on different theories have been built by scholars currently. However, few are refined to a provincial/regional level except for Chinatax^{RCP}.

MFs in Chinatax^{RCP} were compared with those estimated by other methods (Fig.2(b) and Fig.S1). Results indicate that MFs from models applied in EU/OECD countries are mostly higher than those applied in the world and China, whilst MFs in Chinatax^{RCP} are relatively closer to those in models applied in the world. Secondly, monetization results based on damage costs are mostly higher than those based on abatement costs. Thirdly, models that include a wide scope would result in a relatively higher monetization value. Taking PMFP as an example, MFs of PMFP are ranked as EPS, EP, Stepwise, MMG, Ecovalue, ReCiPe, EVR, LIME3, and Chinatax^{RCP} [17, 19,

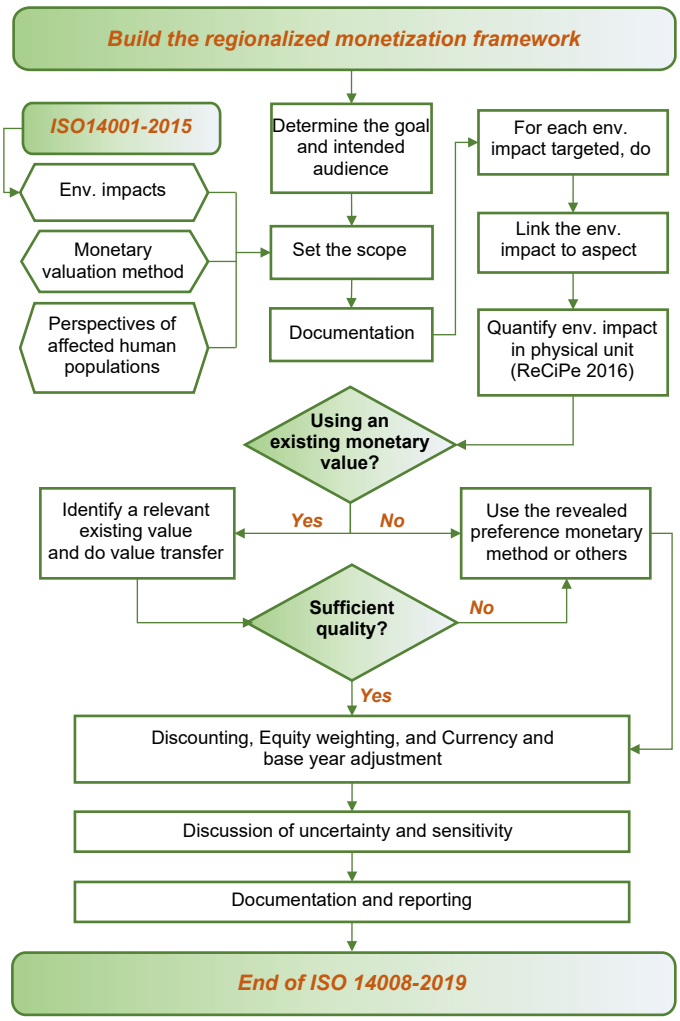


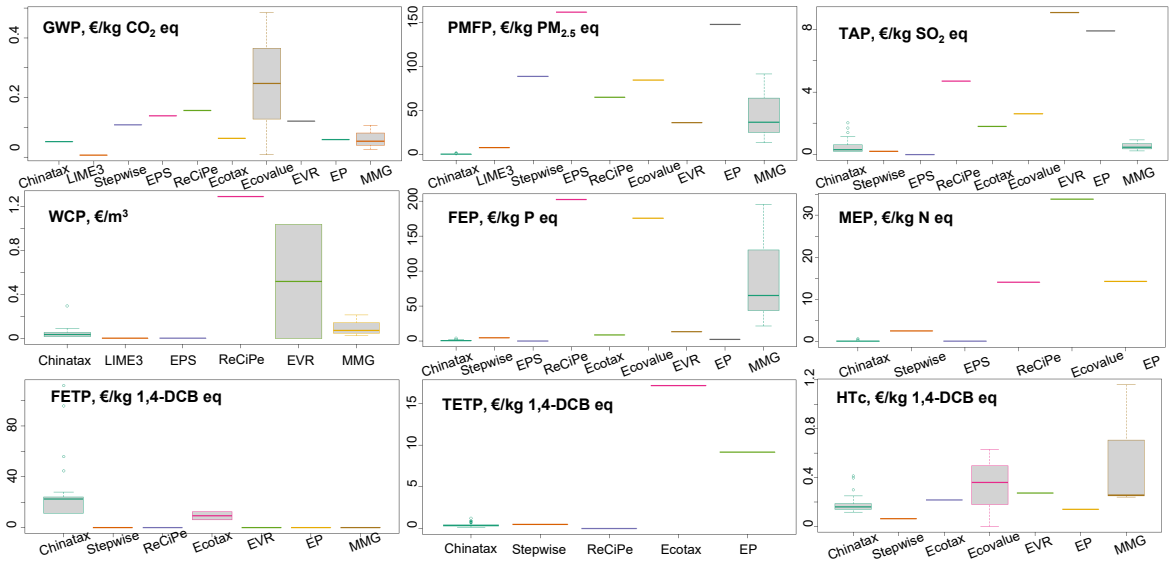
Fig. 1. Flow chart on an application of ISO 14001-2015 and ISO 14008-2019 on building the regionalized monetization framework

25-30]. This is because all the models are based on damage cost except for the lower-ranking EVR and Chinatax^{RCP}. The top seven ranked models are applicable for EU countries except for the globally applicable Stepwise, which is estimated with a wide scope by including the reduced working capacity of human health damage. Though with the same theory, the MF of PMFP in Chinatax^{RCP} is still lower than that in EVR because of their different economic level in EU and China. Despite information mentioned above, not all MFs based on damage costs theory from other countries are higher than those estimated by Chinatax^{RCP}, for example, ozone formation, freshwater/marine/terrestrial ecotoxicity, water use, and terrestrial acidification. This indicates that technological innovation should be encouraged by the Chinese government to reduce the abatement costs of photochemical oxides, chlorobenzene, and SO_x. More attention has been paid to governing the toxicity, acidification, as well as water use in China than those abroad.

a)

Monetization method	Theory	Geographic	Equity weighting	Value transfer	Uncertainty	References
Chinatax[RCP]	Societies' WTP; averting cost; damage/abatement costs;	China, 31 provinces	Regionally considered	With economy adjustments	Qualitatively addressed	Released in this study
EPS	Damage costs, mostly market price	OECD countries	Without regionally considered	-		[26]
Stepwise	Damage costs; budget constraint	Global		-		[27]
LIME3	Damage costs; WTP; conjoint analysis	G20 countries	Globally considered	-	-	[20]
ReCiPe	Damage/abaement/ restoration costs	Global/OECD countries	Without regionally considered	meta-analysis partially	Qualitatively addressed	[30]
Ecotax	Societies' WTP	Sweden		-	Min and max value provided	[21]
EVR (version 1.6)	Abatement costs	Europe, Netherlands		-	Not specifically addressed	[31]
Ecovalue	Marginal abatement costs; WTP; contingent valuation	Sweden		-	Min and max value provided	[29]
Environmental Prices (EP)	Damage costs; abatement costs	Europe		-	Not specifically addressed	[18]
MMG-Method	Damage/abatement/ restoration costs	Europe, Flanders		-	High/middle/low values provided	[28]

b)



c)

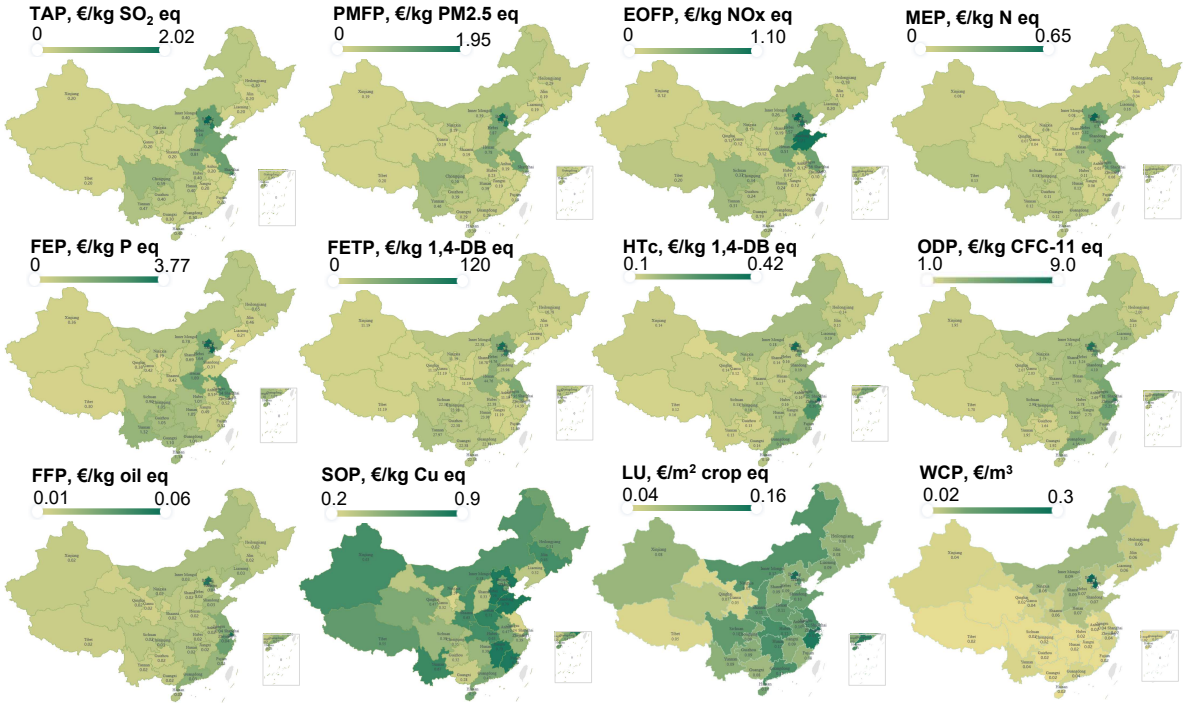
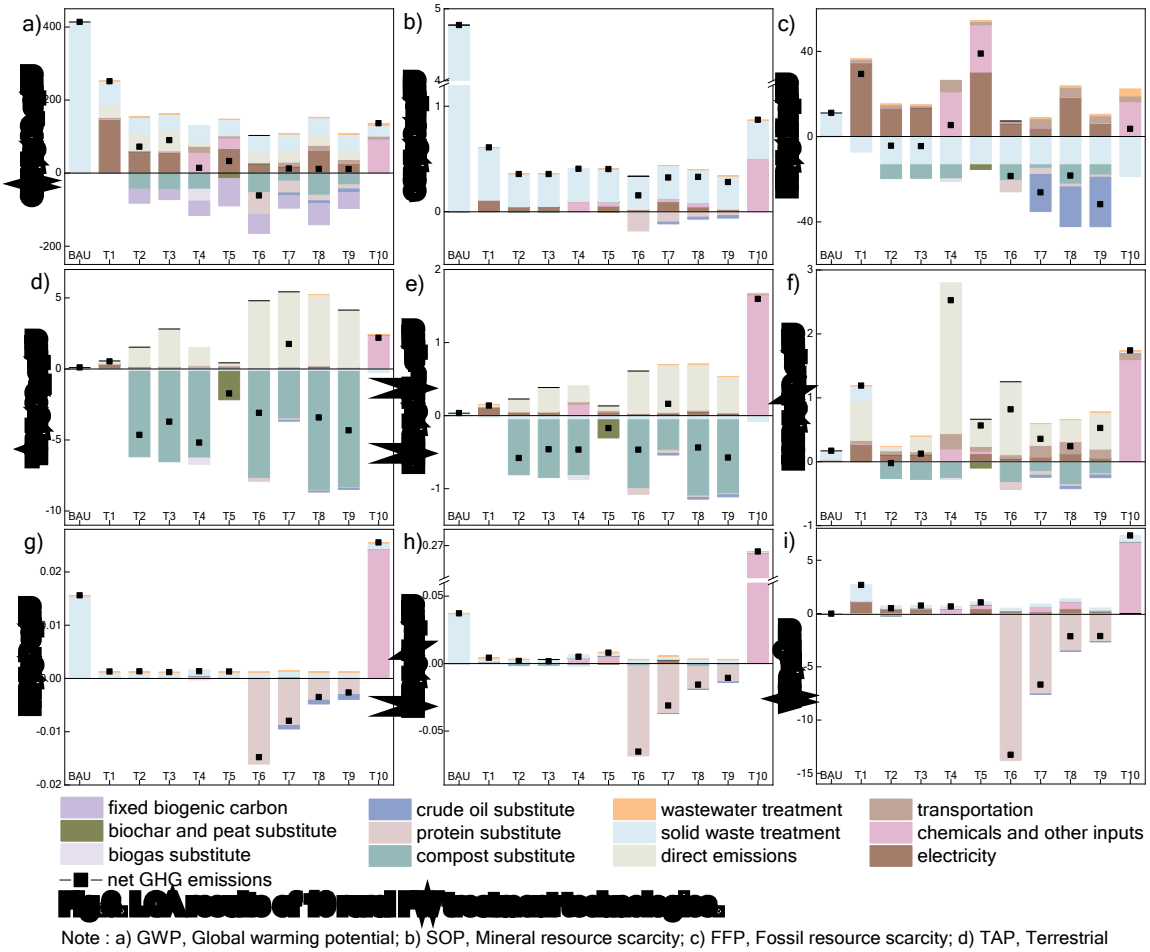


Fig. 2. Comparison of (a) worldwide monetization methods, (b) monetization factors in China and abroad, (c) MFs in 31 provinces in China (GWP, global warming potential; PMFP, fine particulate matter formation; TAP, terrestrial acidification; WCP, water consumption; FEP, freshwater eutrophication; MEP, marine eutrophication; FETP, freshwater ecotoxicity; TETP, terrestrial ecotoxicity; HTc, human health-cancer; EOFP, ozone formation (ecosystem); ODP, stratos. ozone depletion; FFP, fossil resource scarcity; SOP, mineral resource scarcity; LU, land use)

There is an insignificant difference in MFs between Chinatax^{RCP} and the globally applicable models, such as Stepwise, and LIME3 for most of their MFs because the budget constraint method is used for monetization for Stepwise and global differences were considered in LIME3, for example, GWP, ozone formation, FFP, and LU shown in Fig.2(b) and Fig.S1. However, the applicability of Stepwise and LIME3 in China need to be reconsidered giving a mix use of theory by Stepwise and limited data used for conjoint analysis in LIME3.

On the other hand, each MF in Chinatax^{RCP} was regionally/provincially compared (Fig. 2(c) and Fig.S2). Results demonstrated that most monetization values for China's eastern or southeastern areas are higher than those in other areas due to the differences in socio-economic status reflected by per capita income, except for LU, WCP, and IR, which are also affected by biodiversity, water scarcity, radiation levels in different areas in China, respectively.

Environmental impacts of rural FW-to-resources technologies. Ten rural FW-to-resources technologies were assessed via LCA with their system boundaries described in SI Fig. S3 and S7. Comprehensively, biotransformation technologies (T6–T9) achieved environmental savings in 11–14 out of 18 impact indicators, followed by T2–T5 (5–8 out of 18 indicators) (Table S1). Other technologies presented the worst because of producing less qualified products. These results are expected since multiple products via T6–T9 could offset more environmental burdens than single product options (T2–T5). Contribution analysis in Fig. 3 and Fig.S4 illustrates the main drivers of environmental burdens/credits for LCA indicators. T2–T9 resulted in -61.7 to 90 kg CO_2-eq/FU, reducing GHG emissions by over 4 times that of the BAU. T6–T9 reduced the most GHG because of multiproduct substitution. In addition, biogenic carbon was fixed for T2–T9 (Fig. 3a). From the GHG mitigation perspective, as raised in SDG No.13 (Climate Action), bioconversion technologies are the most suitable, which is different from that for the urban, i.e., AD [9, 31]. SOP, an indicator of resource depletion, was lower for rural FW-to-resources technologies than for BAU because less waste is landfill disposed of, indicating resource savings (Fig. 3b). FFP for T6–T9 was the lowest because of crude oil/protein offsets, while FFP for T1 and T5 were the largest, mainly due to electricity use (Fig. 3c). According to SDG 12 (Responsible consumption and production), technologies that minimize energy and resource use whilst maximize the waste reuse are better choices, i.e., T6, T7, and T9. For air-related indicators (i.e., PMFP, HOFP, TAP in Fig. 3d–3f, direct emissions were the main contributors. T2–T9 presented lower environmental results for TAP and PMFP, except T7, due to producing the least compost. T4 contributed the largest share for HOFP mainly due to NMVOCs and NO_x emitted. As noted in SDG 12, emissions to air, water, and soil over the life cycle of a product should be reduced significantly to minimize adverse impacts and maximize reuse, suggesting that T2 is the best choice with negligible emissions and a higher compost yield. For water-related indicators (i.e., WCP, FEP, and MEP) shown in Fig. 3g–3i, T6–T9 demonstrated the lowest values due to BSF/RHF substituting protein from water-intensive cultivated soybean. Amongst, T6 performed better because T7–T9 consumed more electricity to extract oil while producing less compost.



Note : a) GWP, Global warming potential; b) SOP, Mineral resource scarcity; c) FFP, Fossil resource scarcity; d) TAP, Terrestrial acidification; e) PMFP, Fine particulate matter formation; f) HOFP, Photochemical ozone formation to human health; g) FEP, Freshwater eutrophication; h) MEP, Marine eutrophication; i) WCP, Water consumption. BAU, business as usual (landfill); T1, Mechanical drying; T2, Biodrying and maturity; T3, Solar-assisted composting; T4, Underground anaerobic digestion; T5, Heat pyrolysis carbonization; T6, Biotransformation for BSF; T7, Biotransformation for BSF and biooil; T8, Biotransformation for RHF and biooil; T9, dry heat hydrolysis and biotransformation; T10, Enzyme production.)

Chinatax^{RCP} serves for inequality/risk analysis of externality transmedia migration in technical transition. Transmedia migration of environmental impact of the technologies is quantified and visualized in Fig. 4. Total externality costs of T1–T9 (FW reuse options) are lower than those of BAU (landfill option) because most pollutants for BAU would transfer to soil with time (Fig. 4a). This means rural FW reuse could realize overall environmental benefits except for T10 (T10 need to be improved before application). Externality costs for T6–T9 are the smallest because substitution of soybean meal production could reduce environmental burdens, resource, and land consumption, especially improving the land and water quality, which coincides with the viewpoint of Naylor's study [32]. Specific pollutant migration for each medium and resource is shown in Fig. 4b–4f. If technologies of rural FW treatment are transformed from BAU to reuse, comprehensive life-cycle externality costs would be reduced for air quality, soil quality (except T1), water quality (except T3–T4), and resource depletion. However, human health for T1–T9 should be highly concern. This is because process emissions (such as NO_x, and heavy metals) and potential lower environmental credits from the product for some technologies may lead to higher environmental impacts on ozone formation and human health, which added to a higher externality

of human health. By estimation, environmental risks of human health by technology transition of T1 to T9 are 0.4 to 4 times that of BAU. This means that environmental risk caused by a transmedia migration of pollution from "air-water-soil" to "human" should be highlighted during the technical transformation in rural FW management.

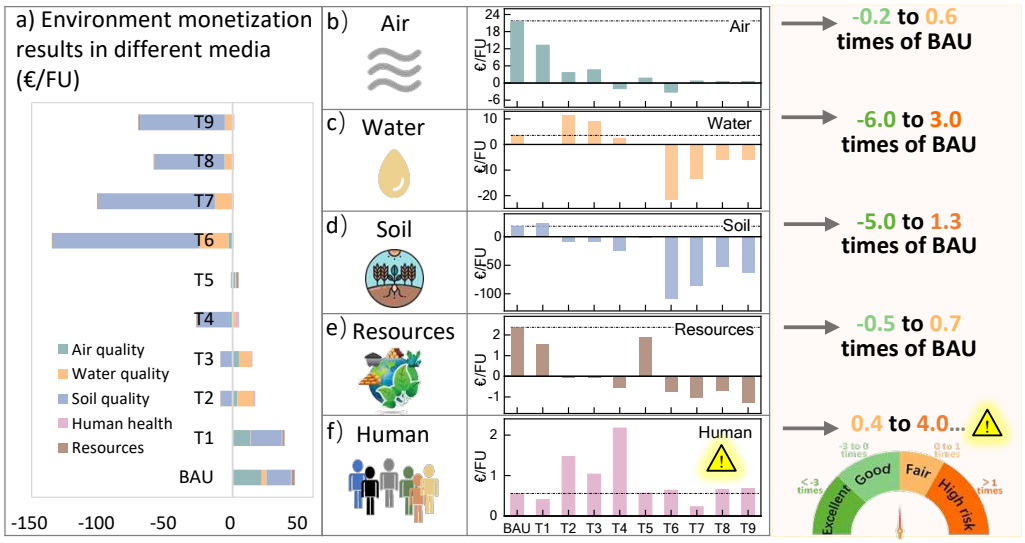


Fig.4. Potential transmedia migration of pollutants in "air-water-soil-human" as well as resources by monetization results. (Note: FU, functional unit; BAU, business as usual (landfill); T1, Mechanical drying; T2, Biodrying & maturity technology; T3, Solar-assisted composting; T4, Underground anaerobic digestion; T5, Heat pyrolysis carbonization; T6, Bioconversion for BSF; T7, Bioconversion for BSF and biooil; T8, Bioconversion for RHF and biooil; T9, Dry heat hydrolysis and bioconversion.)

Economic performances of rural FW-to-resources technologies. The economic analysis of rural FW treatment technologies is presented in Fig. 5, and SD-sheet 5. T2–T4, T6, and T9 could break even in 10 years. Others are unprofitable because of a lower product quality or product yield. Costs and revenues of the technologies were assessed (Fig. 5a to 5c). Results demonstrated that capital costs of T1–T10 fluctuated from 73.6 to 271.6 €/ FU and their potential economic benefits mainly rely on FW disposal fee, which is in line with other studies [31, 33] except for T6 being benefited mainly from product revenue. In the operational stage, (Fig. 5c), T1–T9 fluctuated within 50% of an average operational cost (26.75 €/FU) and an average product revenue (22.49 €/FU), which were both lower than the values for processing urban FW (32.55 € /FU and 33.86 €/FU, respectively) [31]. Land occupation fees should be considered for T6–T9 because they occupy more space for hatching, pretreatment, protein production, and frass composting. These results can serve as a valuable practical guide for rural FW management and deep processing of waste sorting policies in rural China in the future. Details are present in SI.

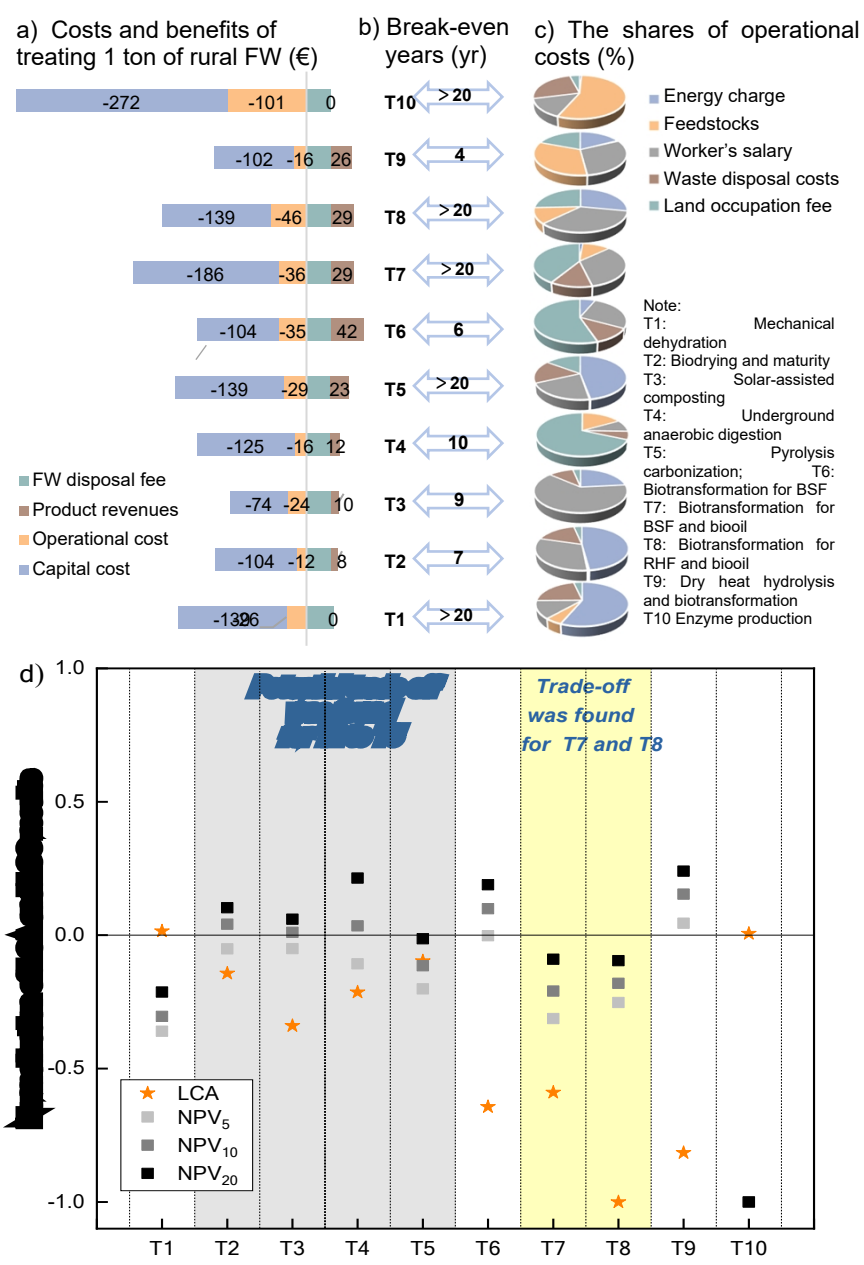


Fig.5. Economic analysis and potential tradeoffs of environmental-economic analysis results of rural FW reuse technologies (a, costs and benefits of processing 1 ton of rural FW; b, break-even years; c, the shares of operational costs; d, potential tradeoffs between E-E analysis results).

Tradeoffs between environmental-economic (E-E) performances in rural FW management. Potential conflicts/tradeoffs between the E-E analysis results for the ten technologies were analyzed by normalizing both performances (Fig. 5d and SD-sheet 7). Results showed that tradeoff exists for T7–T8 because they presented environmental savings but negative economic benefits. In addition, potential trade-offs are found for T2–T5. Detailed explanations and information are in SI. These trade-offs may impede technology selection and policy/decision making more sustainably.

Chinatax^{RCP} serves for environmental-economic integration analysis in sustainable-technological transition. Chinatax^{RCP} , a regionalized life-cycle monetization model built in this study, could handle the conflicts by internalizing externalities [10]. The net present values (NPVs) and break-even years of ten FW reuse technologies before and after E-E integration are shown in Table S2. After E-E integration, T3 was unprofitable. Environmental externalities have little impact on break-even years for T2 and a positive impact on T4 compared to others. Although T7–T9 presented worse economic performances, environmental externality costs have the potential to offset their economic loss. Comprehensively, T2, T4, and T6–T9 are preferable from an environmental perspective by the government. Considering that investment and simple operation lead first for enterprises, especially in rural areas, T2, T6, and T9 are suggested for enterprises under the financial support of local government.

Substitution and policy scenario analysis. According to Fig. 6-S1, protein substitution with soybean meal performed better than substitution with fish meal mainly because of inputs required for soybean planting can be offset while there is little input during fish meal production, which is consistent with the viewpoint of Naylor et al. [32]. Different substitutes for biogas and compost resulted in an insignificant impact on the final environmental impact monetization (EIM) result, as seen in Fig. 6-S2 and S3, except for S3-VII (a variation of 44% with an NPK substitution ratio of 1:1:1). If biochar is substituted by coal, its EIM result would be negative (-36.1 €/t rural FW) (Fig. 6-S4). Fig. 6-S5 represents the scenarios under different FW disposal fees. Results indicated that T1, T5, T7, T8, and T10 could not be profitable even with high subsidies, i.e., 30.5 €/t FW, whilst others could break even. When the Chinese average FW disposal fee (14.0 €/t FW) was provided, NPV_{20} for only T6 and T9 were positive. No technology could survive under zero subsidies, indicating that the rural FW treatment/reuse system relies heavily on government funding, similar to views held by others [4, 34]. Scenarios with/without environmental governances for T9 are demonstrated in Fig. 6-S6. The net income of T9 was found to be higher than that of the conventional BAU. Under no environmental governance for T9, environmental benefits from technology transition, i.e., BAU to T9, were positive (approximately 115.7 €/t rural FW), meaning that a potential weak environmental governance is not an obstacle to technology transition for T9 in rural areas. On the other hand, the environmental benefits of emission reduction could not offset the economic costs of emission reduction. Given this, we recommend that the local government spare extra waste government subsidies (i.e., 5.87 €/FU for T9) support for secondary waste governance in rural FW management in decoupling FW disposal fees with rural FW management.

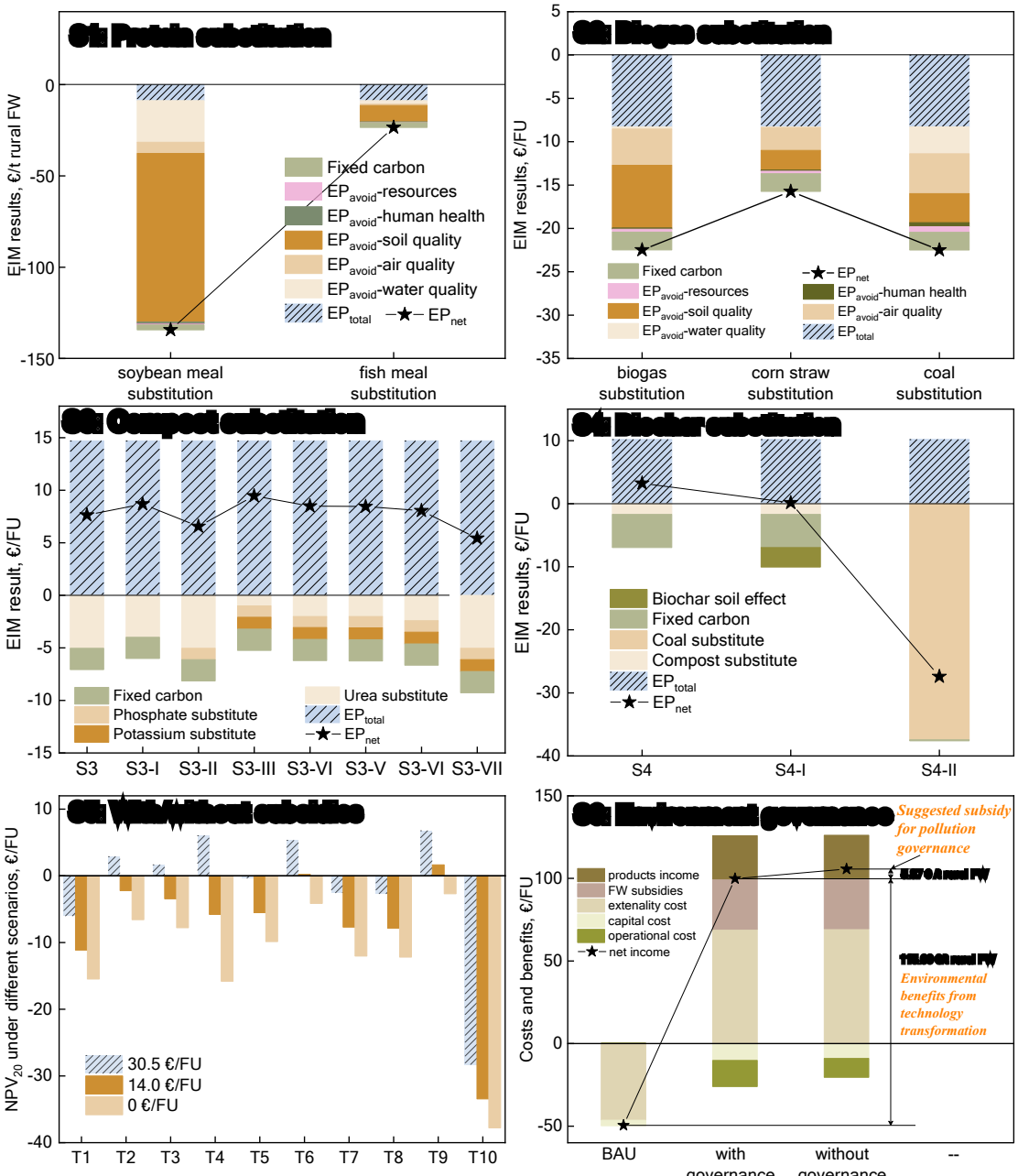


Fig.6. Scenario analysis of monetization results with different substitution methods and environmental governance measures.

S1: Monetization results comparion of protein substitution with soybean meals and fishmeal (T9).
S2: Monetization results comparison of biogas substitution with corn straw and coal (T4).
S3: Compost substitution with chemical fertilizer for T2 in different NPK ratios shown in Table.S3.
S4: Monetization results comparison of biochar substitution with compost (with/without biochar soil effect) or coal (T5).
S5: Environment and economic comparison under the context of different environmental governance conditions (T9).
S6: Costs and benefits comparison with/without FW treatment subsidies (T9).

Discussion and policy implications

Technologies' transition from traditional to novel options may accompany tradeoffs among different environmental issues, and among environmental, economic, and/or other issues, as well as pose inequities and risks by transmedia/transregional migration of environmental externalities [35, 36]. The development of life-cycle monetization methodologies can serve for ensuring a sustainable technology transition which is of great importance in tackling climate change issues

and promoting sustainable development [37, 38]. Given the circumstances that a regionalized monetization method in China is lacking, Chinatax^{RCP} is constructed to assist decision makings by policymakers and entrepreneurs.

Compared with the worldwide monetization methods, we found that our monetization factor results are closer to those with global scope. Regional Chinatax^{RCP} monetization factors demonstrate that those for China's eastern or southeastern areas are higher than those in other areas due to advances in their economies.

As a case study, we performed the environmental-economic integration analysis of rural FW management in Zhejiang province with Chinatax^{RCP} by considering externalities. Results indicated that biodrying & maturity process (T2) and bioconversion technologies (T6 and T9) could be potential candidates owing to environmental credits brought by protein and/or compost replacing soybean meal and/or mineral fertilizer. These conclusions are different from studies focusing on urban waste which was more preferred to be treated via centralized AD (Table S3). With regards to the potential tradeoffs among environmental impacts, transmedia ("air-water-soil-human") migration and resource consumption of environmental externalities using Chinatax^{RCP} were performed to visualize potential environmental risks in technological-sustainable transition. Results indicated that overall environmental pollution in air-soil-water media and resource depletion were relieved but at the cost of human health. This enlightened us that effective control for direct emissions should be emphasized in rural FW management because they are the main and direct cause of human health risks.

Product substitution and policy changes would significantly affect strategies of technical transition [11, 39, 40] (Table S3). For example, soybean meals and coal were more effective than others for substitution by larvae and biochar products, respectively. Negligible credit of biochar used as soil conditioner was found, which is inconsistent with the result by Yang, mainly because of the lower yields in our study [41]. This inspires us to pay more attention to the effective use of products, instead of emphasizing cleaner production only. Results of policy scenarios indicate that FW disposal fees are vital for rural FW valorization. Given this, we recommend extra waste subsidies to control emissions in the decoupling way of rural FW management with subsidies, to stimulate/encourage emission control caused by rural FW management.

Overall, our study established a quantitative, comprehensive, and regionalized externalities' monetization model in China for the first time to our best knowledge. It was applied to evaluate ten rural FW waste-to-resources technologies as part of a sustainable technical transition in the food sector, demonstrating its capability to support decision making from an integrated environmental-economic perspective. However, as an externalities' monetization method based on Recipe 2016, Chinatax^{RCP} can be used in a wider context in supporting the development of a sustainable society.

Methods

Establishment of a regionalized life-cycle monetization model in 31 provinces in China. Chinatax^{RCP} is a regionalized life-cycle monetization model based on regional environmental protection taxes, resource taxes, and carbon abatement costs in China (SD-sheet 1 to 16). Based on the principle of compensating for the environmental abatement cost and the sewage charging standards in the past years, the environmental protection tax also takes into account the environmental carrying capacity for each pollutant, the unit cost of pollution control of local enterprises, the level of regional economic development, the basic quality of the ecological environment in each regional, the discharge status of pollutants, as well as the requirements of ecological development goals in China. Chinatax^{RCP} is established to handle the potential tradeoffs between E-E analysis results, to reflect the transmedia migration of pollutants (air-water-soil-human health) and resource depletion, as well as to present the government's willingness to pay for environmental governance. Regarding resource depletion, the purpose of resource tax is to adjust the differential income of resources, to improve the efficiency of resource exploitation, to promote fair competition, and to reflect the willingness to pay for resource consumption. Therefore, resource taxes in 31 provinces are used for the correction of the resource-related indicator. In this study, Per capita income for each province is also used as one of the modification factors. MFs of 18 LCA midpoint indicators in ReCiPe 2016(H) in 31 Provinces in China were estimated according to the information below.

GWP. The average MF for GWP (approximately 0.052 € /kg CO_2 eq) is estimated by the industrial abatement costs of CO_2 in six major carbon emission sectors in China, i.e., construction, cement, power, iron & steel industry, and petrochemical industry (Data is derived from the Ministry of Ecology and Environment in China, and references are shown in Supplementary Dataset).

$$w_{GWP,\ average} = \frac{\sum_{t=1}^{t=6}(\mathrm{AC}_t \times \mathrm{AP}_t)}{\sum_{t=1}^{t=6}\mathrm{AP}_t} \quad (1)$$

AC_t: The amount of CO_2 abated in sector t (including 6 major sectors, i.e., construction, cement, power, iron and steel industry, and petrochemical industry sector).

AP_t: The CO_2 abatement cost in sector t including 6 sectors mentioned above.

$w_{GWP,\ average}$: The average MF for GWP in China.

PMFP, FEP, MEP, EOFP, HOFP, and TAP. MFs of these six indicators in 31 provinces are estimated according to Formulas (2)–(4) considering that each of their environmental indicators are caused by multiple environmental pollutants [42]. Data for the pollutant amounts in each province $a_{i,j,m}$ and pollution equivalent $b_{i,j,m}$ are collected according to information from the National Bureau of Statistics and the National Bureau of Environmental Protection. The influence potential coefficient $f_{i,j,m}$ represents the relevant potential influence proportion per unit of

pollution i to indicator j. Applicable taxes of emissions $r_{i,j,m}$ are investigated according to references in Supplementary Dataset.

$$c_{i,j,m} = \frac{1}{b_{i,j,m}} \times r_{i,j,m} \quad (2)$$

$$e_{i,j,m} = \frac{\mathrm{f}_{i,j,m} \times a_{i,j,m}}{\sum(\mathrm{f}_{i,j,m} \times a_{i,j,m})} \quad (3)$$

$$w_{j,\ m} = \sum(e_{i,j,m} \times c_{i,j,m}) \quad (4)$$

$a_{i,j,m}$: Average emissions amount in 2020 for pollutant i related to indicator j in province m.

$b_{i,j,m}$: Pollutant equivalent of the emissions i related to indicator j in province m.

f_{ij}: Influence potential coefficients per unit of pollution i for LCA indicator j in province m.

$r_{i,j,m}$: Applicable tax of emissions i in 2018 for indicator j province m, yuan (finally exchange to Euro) per pollutant equivalent.

$c_{i,j,m}$: Monetary adjustment factor for indicator j related to pollutant i in province m.

$e_{i,j,m}$: Influence potential coefficient of pollution i for LCA indicator j in province m.

$w_{j,m}$: MF of indicator j.

d: Discounting rate, i.e., 3%.

The impacts of $PM_{2.5}$ are found to be approximately four times as harming as PM_{10} [16, 43]. Monetary adjustment factor of PMFP calculated by PM_{10} were converted to $PM_{2.5}$ by multiplying by 4.

Ecotoxicity (FETP, METP, and TETP). The indicators, i.e., FETP, and METP are calculated by Formula (5) to (7) using monetary adjustment factors, midpoint characterization factor for LCA indicators, and coefficient of wastewater to sea/river in each province [44].

$$E_m = \frac{Ex_m}{Average(Ex_m)} \quad (5)$$

$$w_{FETP,\ m} = \frac{1}{b_{i,j,m}} \times r_{i,j,m} \times E_m \quad (6)$$

$$w_{METP,\ m} = w_{FETP,\ m} \times \frac{SD_{marine}}{SD_{fresh}} \times \frac{CF_{marine}}{CF_{fresh}} \quad (7)$$

E_m: Economic modifying factor for indicator j in province m in China.

Ex_m: Per capita income in province m in China.

SD_{marine}: Species density in marine water, species/m^3 in marine water, i.e., 3.46×10^{-12}

SD_{fresh}: Species density in fresh water, species/m^3 in fresh water, i.e., 7.89×10^{-10}

CF_{marine}: conversion factor from mid to endpoint for METP, species in marine water yr/kg 1,4 DCB eq, i.e., 1.05×10^{-10}.

CF_{fresh}: conversion factor from mid to endpoint for FETP, species in freshwater yr/kg 1,4 DCB eq, i.e., 6.95×10^{-10}.

$w_{METP,m}$: MF for indicator METP in province m in China.

$w_{FETP,m}$: MF for indicator FETP in province m in China.

TETP is estimated according to Stepwise 2006 based on budget constraints of the environmental protection expenditures in developed countries in 2008 (approximately 1-2% of gross domestic product), which is approximate to the current environmental protection expenditures in China in 2020 according to statistics of Ministry of Ecology and Environment of the People's Republic of China. Therefore, MF estimation of TETP in China by Stepwise 2006 is reasonable. Per capita income and fertilizing intensity of phosphate in each province are used to estimate the normalization results of TETP in each province. Formulas are shown as (8) to (11).

$$Intense_m = \frac{\mathrm{Ph}_{amount}}{\mathrm{S}_m} \times \frac{31 \times 2}{142} \tag{8}$$

$$MFR_{intensity,\ m} = \frac{Intense_m}{average\ (Intense_m)} \tag{9}$$

$$w'_{TETP,m} = \frac{\mathrm{CPI}_{EU,\ 2019}}{\mathrm{CPI}_{EU,\ 2020}} \times \frac{\mathrm{PPP}_{China,\ 2020}}{\mathrm{PPP}_{German,\ 2020}} \tag{10}$$

$$w_{TETP,m} = \sum w'_{TETP,m} \times MFR_{intensity,\ m} \times E_m \tag{11}$$

Ph_{amount}: The application amount of phosphate in province m

S_m: Total sown area of crops in province m

$MFR_{intensity,\ m}$: Modification factor of fertilization intensity in province m.

$Intense_m$: Application intensity of phosphorus in province m.

$w'_{TETP,m}$: The average MF of TETP in China.

$w_{TETP,m}$: MF of TETP in province m in China.

$\mathrm{CPI}_{c,\ n}$: Consumer price index in country c in year n.

$\mathrm{PPP}_{c,\ n}$: Purchasing power parity in country c in year n.

Human toxicity (cancer or noncancer). $\mathrm{HT_c}$ and $\mathrm{HT_{nc}}$ are estimated with the value of life year of Chinese residents in 2016 (5.54×10^4€/DALY), converted to kg 1,4 DB eq using factors in Usetox model from Simapro software, and customized to factors for 31 provinces in China using per capita income. Formulas (12) to (13) are as below.

$$V_{2020} = \frac{\mathrm{CPI}_{2020}}{\mathrm{CPI}_{2018}} \times V_{2018} \tag{12}$$

$$w_{HT_{j,m}} = \sum V_{2020} \times CF_{(non)cancer} \times E_{j,\ m} \tag{13}$$

V_n: VLY of Chinese residents in the year n, i.e., WTP value in per DALY.

$CF_{(non)cancer}$: Conversion factor from DALY to kg 1,4 DCB eq, $conf_{cancer}$ equals to 3.32×10^{-6}, and $conf_{noncancer}$ equals to 2.28×10^{-7} according to CF in Usetox model.

$E_{j,m,}$: Economic modifying factor for indicator j in province m in China.

CPI_n: Consumer price index in China in year n.

Ionizing radiation (IR). Considering that MFs in Stepwise and EPS are globally applicable. MF of ionizing radiation (IR) is estimated with the average IR value from Stepwise and EPS in 2020,

and customized to MFs in 31 provinces using per capita income, air absorbed dose rate, and radiation concentration in aerosol and deposition [45]. Formula can be seen below.

$$MFR_m = \frac{DR_m}{Average(DR_m)} \tag{14}$$

$$w_{IR,\ m} = IR_{Ecotax} \times \frac{\mathrm{CPI}'_{EU,\ 2020}}{\mathrm{CPI}'_{EU,\ 2019}} \times \frac{\mathrm{PPP}_{China,\ 2020}}{\mathrm{PPP}_{EU,\ 2020}} \times MFR_m \times E_m \tag{15}$$

IR_{Ecotax}: The average MF of IR in Ecotax 2006 in Switzerland, €/kBq Co-60 to air

$\mathrm{CPI}'_{c,\ n}$: Consumer price index in Europe in year n.

DR_m: Air absorption dose rate of γ-ray in province m, nGy/h.

MFR_m: Modification factor of radiation in province m.

ODP. The MF of ODP is estimated according to the total investment of Ozone Depleting Substances (ODS) abatement (USD 53.64 million in 2006) and the total reduction amount of ODS (25,000 tons) in 2006 [42, 46], which is normalized to a factor for Zhejiang using the adjustment factor of ozone concentration and per capita income in China.

$$w_{ODP,\ 2006} = \frac{Invest_{total}}{ODS_{total}} \tag{16}$$

$$MFO_{ozone,\ m} = \frac{OC_{ozone,\ m}}{Average(OC_{ozone,\ m})} \tag{17}$$

$$w_{ODP,\ 2020,\ m} = E_m \times MFO_{ozone} \times \frac{\mathrm{CPI}_{2020}}{\mathrm{CPI}_{2006}} \tag{18}$$

$Invest_{total}$: Total investment of ODS abatement, USD

ODS_{total}: The total reduction amount of ODS, tons

$MFO_{ozone,\ m}$: Modification factor of ODP in province m.

$OC_{ozone,\ m}$: The average ozone concentration (the average maximum daily ozone concentration at the 90th percentile for 8 hours) in province m in 2020, μg/m^3.

$w_{ODP,\ 2006}$: MF of ODP in year n.

FFP and SOP. MF for FFP and mineral resource scarcity (SOP) in 31 provinces are estimated according to resource taxes of Cu and oil in each province and trading price of Cu and mineral oil in Chinese market. Oil_{price} has varied significantly under the context of COVID-2019. To eliminate its influence on the real market, a 10-year average oil price was used. Relevant references are shown in Supplementary Dataset. Formula (19) to (20) can be seen below.

$$w_{FFP,\ m} = \mathrm{Average}\ (Cu_{price,\ 2020}) \times Cu_{tax,\ m} \tag{19}$$

$$w_{SOP,\ m} = \mathrm{Average}\ (Oil_{price,\ 2011-2020}) \times Oil_{tax,\ m} \tag{20}$$

$Cu_{tax,\ m}$: Copper tax of mineral processing in province m.

$Oil_{tax,\ m}$: Oil tax in province m.

WCP and LU. The average resource fee or tax of surface water in each province are used as the MF of the indicator WCP in 31provinces. Currently, both land use tax and tax on cultivated land in China are inappropriate to estimate MF in each province [47]. Because the levying purpose of

land use tax is to: improve land use efficiency and land circulation, adjust land income distribution, regulate land tenure, promote income sharing, and optimize land use structure, not only to reflect the environmental damage/governing costs in land use. In this model, MFs of land use and land transformation in the ReCiPe monetization model mentioned in Fig.1(a) were used and transferred as MF of LU in China. Biodiversity damage costs were used as the baseline theory to merge MF of land use and land transformation since both are included in the LU indicator in ReCiPe 2016. The obtained MF of LU was normalized to 31 provinces by using the tax on cultivated land occupancy. Estimation details are shown in formulas (21) to (29).

$$MSA_{f\to c} = \frac{(S_f - S_c)}{S_f} \tag{21}$$

$$MSA_{g\to c} = \frac{(S_g - S_c)}{S_g} \tag{22}$$

$$MFR_{CPI} = \frac{\mathrm{CPI}_{EU,\ 2021}}{\mathrm{CPI}_{EU,\ 2020}} \tag{23}$$

$$w'_{LU,\ f,\ 2020} = P_f \times MSA_{f\to c} \times \frac{w'_{LU,\ f,\ 2021}}{10000} \times MFR_{CPI} \tag{24}$$

$$w'_{LU,\ g,\ 2020} = P_g \times MSA_{g\to c} \times \frac{w'_{LU,\ g,\ 2021}}{10000} \times MFR_{CPI} \tag{25}$$

$$w'_{LT,\ f,\ 2020} = 0.5 \times P_f \times MSA_{f\to c} \times \frac{w'_{LT,\ f,\ 2021}}{10000} \times \frac{1}{t_{rel,\ f}} \times MFR_{CPI} \tag{26}$$

$$w'_{LT,\ g,\ 2020} = 0.5 \times P_g \times MSA_{g\to c} \times \frac{w'_{LT,\ g,\ 2021}}{10000} \times \frac{1}{t_{rel,\ g}} \times MFR_{CPI} \tag{27}$$

$$MFR_{PPP} = \frac{\mathrm{PPP}_{China,\ 2020}}{\mathrm{PPP}_{EU,\ 2020}} \tag{28}$$

$$w_{LU,\ 2020,\ m} = (\sum w'_{LU,\ f,\ 2020} + \sum w'_{LT,\ 2020,\ m}) \times MFR_{PPP} \times MFR_{lut,\ m} \tag{29}$$

P_f: The proportion of forest biomes to the total global terrestrial areas, i.e., 40% [48].

P_g: The proportion of grassland biomes to the total global terrestrial areas, i.e., 60% [48].

$w'_{LU,\ 2021}$: MF of LU by land type forest or grassland in True Price model

$w'_{LT,\ 2021}$: MF of land transformation in True Price model

$w'_{LU,\ 2020}$: The average MF of LU in China

$w_{LU,\ 2020,\ m}$: MF of LU in China in province m

$MSA_{f\to c}$: Mean species abundance when forest transferred as cropland.

$MSA_{g\to c}$: Mean species abundance when grassland transferred as cropland.

S_f: relative species richness of forest, i.e., 1

S_g: relative species richness of grassland, i.e., 0.67

S_c: relative species richness of annual crops, i.e., 1

$MFR_{lut,\ m}$: Modification factor of land use tax in province m.

462 BDF_c: BDF of annual crops, i.e., 0.6 [49].

463 BDF_g: BDF of grassland or pasture or meadow, i.e., 0.33 [49].

464 **Technology description, goal, scope, and assumptions.** Ten technologies demonstrated
465 in Zhejiang Province were investigated for a comparative study with BAU-landfill disposal, as
466 Zhejiang explored more routes in the field of rural FW governance [33, 50]. Descriptions of these
467 FW-to-resources technologies can be found in our newly published book [33] and Supplementary
468 information (SI). The goal of this study is to evaluate both the E-E performances of ten rural FW
469 reuse technologies and to compare them with the conventional BAU-landfill route. The functional
470 unit is processing 1 ton of rural FW. The system boundary begins with the transport of the
471 collected rural FW to the disposal/use of final products. The system includes the production of
472 other raw materials used in all unit processes, transport, processing, product use, and waste
473 discard. Final products generated in rural FW management can replace the traditional and similar
474 commodities on the market (Fig. S7). Assumptions are shown in Supplementary Dataset (SD)-
475 sheet 0. Material and carbon flow analyses (Fig. S3) were applied to track materials and carbon
476 fate in rural FW management.

477 **LCA and LCC analysis.** The life cycle inventories of technologies are presented in Table S4.
478 Detailed information is shown in SI. LCA was performed by GaBi software. 18 LCA categories in
479 ReCiPe 2016 were assessed. To analyze the economic feasibility of rural FW management
480 technologies, life-cycle costing (LCC) was performed with results indicated by net present value
481 (NPV) and break-even years. The $NPV_{5,\ 10}$ and LCA results of treating per ton of rural FW were
482 normalized to reveal potential trade-offs between LCA and LCC results. Detailed information on
483 indicators is presented in SI. Environmental burdens in this study are from raw materials, energy,
484 and waste discharge. Whilst environmental credits mainly result from the substitution of
485 conventional products. The net environmental impact results are equal to the differences between
486 environmental burdens and credits (SD-sheet 17 to sheet 20).

487 **Tradeoffs between environmental and economic performances.** Potential
488 conflicts/tradeoffs between the E-E analysis results for the ten technologies were analyzed by
489 normalizing both results (SD-sheet 21). The positive values of NPV represent technical profit and
490 vice versa. The higher values of LCA results indicate their environmental impacts are more
491 severe. Trade-offs exist when the preference from either environmental or economic results
492 conflict.

493 **Transmedia migration analysis of environmental externality in technical transition.**
494 The built regionalized life-cycle monetization framework was used to estimate the externality
495 costs of each LCA indicator. The obtained life-cycle externality costs of 18 LCA mid-point
496 indicators were grouped to five categories for the ease of result visualization, according to
497 receiving media of emissions or those being affected, i.e. water quality related (including WCP,
498 FETP, FEP, METP, MEP), air quality-related (including LCA indicators GWP, PMFP, EOFP,
499 ODP), soil quality related (including LCA indicators cultivated land use, TAP, TETP), human

health related (including HT_c, HT_{nc}, IR, HOFP), and resource depletion related (including SOP, FFP) [11, 12] (SD-sheet 22).

Environmental-economic integration by externality costs. In addition to traditional incomes and expenses in LCC, life-cycle externality costs estimated as above were included as an environmental cost to integrate E-E perspectives. NPV and break-even years remained indicators to present the integrated results. E-E integration is shown in SD-sheet 23.

Sensitivity and Scenario analysis. Sensitivity analysis was assessed with a one-at-a-time approach by adjusting 10% of each environmental factor (i.e., inventory feedstock) and economic factor (i.e., product revenue, capital investment, and operational cost) [11], which are presented in SI. Choices of product substitution in FW treatment might affect the comparison results significantly and vary across studies [39, 40]. S1–S4 was set to investigate the influences of protein, biogas, compost, and biochar substitutions on the EIM results. Detailed information is described in SI, Table S5, Table S6, and SD-sheet 25. Treatment technologies selected for scenario studies and the baseline substitution or policy scenario are indicated in brackets:

S1: protein substitution for fishmeal (T9, soybean meal substitution at baseline).

S2: biogas substitution for coal (T4, biogas from manure substitution at baseline).

S3: compost substitution in different NPK ratios (T2, NPK = 0.4:0:0 at baseline).

S4: biochar substitution for compost or coal (T5, without biochar soil effect in baseline).

S5: comparison with different FW treatment subsidies (T1–T10, 30.5 € t/FW at baseline).

S6: comparison under different environmental governance conditions (T9, pollution control is in place in the baseline).

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Data Availability

Detailed information is shown in Supplementary Information. Datasets related to this article can be found at Supplementary Dataset-sheet 1 to sheet 11 summarized below:

Content of Supplementary Datasets	
Sheet 0	Assumptions and prerequisites
Sheet 1	Relevant pollution equivalent
Sheet 2	Applicable tax in 31 provinces
Sheet 3	Monetary adjustment factors
Sheet 4	Emissions in 31 provinces

Sheet 5	Influence potential coefficient
Sheet 6	Water use
Sheet 7	Resources (copper and oil)
Sheet 8	Human toxicity (cancer and noncancer)
Sheet 9	ODP (Ozone layer depletion
Sheet 10	GWP (Global warming potential)
Sheet 11	IR (Ionizing radiation)
Sheet 12	Ecotoxicity (FETP, METP, and TETP)
Sheet 13	Land use
Sheet 14	Monetization factor (MF) results in 31 provinces
Sheet 15	MF in China exchanged as Euro
Sheet 16	MF comparison in home and abroad
Sheet 17	Rural FW-Life cycle inventory
Sheet 18	FW transport
Sheet 19	NPV-excluding externality costs
Sheet 20	LCA results
Sheet 21	Normalization results
Sheet 22	Monetization results
Sheet 23	NPV-including externality costs
Sheet 24	Carbon flow analysis
Sheet 25	Sensitivity analysis (LCA results and LCC results)

CRediT

Fang Liu: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Data curation, Writing–original draft, Visualization. **Liqing Xin**: Validation, Formal analysis, Investigation, Data curation, Revision. **Hang Tang**: Investigation, Data curation, Validation. **Yong Qin**: Resources, Investigation, Data curation. **Lei Zhang**: Investigation, Data curation, Validation. **Xinyue Dong**: Investigation, Validation. **Yi Zhang:** Validation. **Weixiang Wu**: Resources, Validation, Revision, Funding acquisition. **Lei Wang**: Conceptualization, Methodology, Validation, Investigation, Supervision, Resources, Writing–review & editing, Funding acquisition.

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