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Toward climate-smart irrigation: evaluating the sustainability of negative pressure systems through carbon-nitrogen footprint and cost-benefit analysis

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Abstract

In protected agriculture, irrigation methods and input structures play a critical role in shaping carbon and nitrogen footprints as well as ecological-economic performance. Using a life cycle assessment (LCA) approach, this study systematically evaluated the carbon footprint (CF), nitrogen footprint (NF), environmental damage cost (EDC), and net ecosystem economic benefit (NEEB) under conventional surface irrigation (CK) and three negative-pressure irrigation (NPI) treatments (N1, N2, N3). Compared with CK, NPI significantly reduced CF and NF by up to 74.09% and 76.45%, respectively, primarily due to reduced N₂O emissions and fertilizer inputs. NPI also alleviated soil organic carbon loss by 78.85%, underscoring its strong potential for environmental sustainability. However, the high cost of ceramic emitters (accounting for 88.36%–89.80% of Cost_{agricultural input}) and their substantial upstream emissions resulted in a significantly lower net economic benefit (NEB) for NPI compared with CK. Despite this drawback, NPI treatments demonstrated superior ecological-economic efficiency: EDC was reduced by over 85%, and the N1 treatment achieved a NEEB of -10.37 CNY ha⁻¹, nearly reaching break-even. These results highlight a clear trade-off between environmental benefits and economic feasibility. Policy support (e.g., subsidies, carbon credits) and innovations in emitter materials (e.g., low-carbon or biodegradable alternatives) are essential to improve the overall sustainability and scalability of NPI systems.

Keywords: Negative pressure irrigation, life cycle assessment, carbon and nitrogen footprints, environmental-economic trade-offs, greenhouse gas emissions



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INTRODUCTION

With the intensification of agriculture and the growing threat of climate change, enhancing resource-use efficiency while reducing agricultural greenhouse gas (GHG) emissions and reactive nitrogen (Nr) losses has become essential for the green and low-carbon agricultural transformation^[1-4]. In protected vegetable production systems, in particular, high demands for water and fertilizers, intensive input requirements, and environmental sensitivity make traditional irrigation methods, such as furrow and sprinkler irrigation, not only inefficient in water and nutrient use but also major contributors to nitrogen leaching, ammonia volatilization, and N₂O emissions^[5-7]. Together, these processes create substantial carbon (CF) and nitrogen footprints (NF)^[8]. Consequently, the development of efficient and intelligent water-fertilizer management technologies is central to ensuring the sustainability of protected agriculture^[9].

Negative pressure irrigation (NPI) is a precision irrigation technique driven by crop root-zone water demand. Using porous ceramic emitters, it establishes a stable negative pressure that enables self-regulated uptake of water and nutrients^[10-14]. This method significantly reduces unnecessary water and fertilizer application while improving the synchronization of nutrient absorption by crops^[15,16]. Previous studies have demonstrated that NPI can reduce irrigation water use by more than 30%^[13], decrease nitrogen losses by 20%-40%^[17,18], and effectively suppress GHG emissions such as N₂O, all while maintaining crop yields^[19,20]. However, most existing research has focused on the field-level agronomic and environmental benefits of NPI, whereas the upstream environmental costs - particularly those associated with the production and deployment of irrigation infrastructure (e.g., ceramic emitters, pipelines, and control devices) - remain largely underexplored^[21]. Because these components are often manufactured through energy-intensive processes (e.g., high-temperature sintering), their environmental footprints may offset gains achieved during the operation phase. Assessing these upstream impacts is therefore critical to evaluating the overall sustainability of NPI systems, yet they have received limited quantitative attention. This gap hinders a full understanding of the life cycle trade-offs involved in adopting NPI technologies.

Furthermore, current evaluation metrics are often narrow, focusing only on yield^[19], water-use efficiency^[17], or GHG emissions^[20], without integrating upstream resource consumption or accounting for environmental externalities in economic terms. A comprehensive assessment framework that combines CF and NF with economic performance is lacking, hindering evidence-based decision making for policy design and technology adoption.

To address these gaps, this study evaluates the environmental and economic sustainability of NPI in a protected Chinese chive production system. By combining field experiments with a life cycle assessment (LCA) approach, we construct a system boundary encompassing “resource input-production process-environmental output”, enabling the systematic quantification of GHG emissions, Nr loss pathways, and the associated CF and NF across both upstream and downstream stages. We further integrate productivity and environmental impact into a net ecosystem economic benefit (NEEB) model to capture trade-offs between environmental gains and financial viability. The objective is to provide a holistic and quantitative foundation for the adoption and scaling of NPI systems, ultimately supporting the sustainable transformation of protected agriculture.

MATERIALS AND METHODS

Study site and experimental design

The experiment was carried out in a rain shelter at the State Key Laboratory of Hydrology-Water Resources and Hydraulic Engineering in Nanjing, China (32.06° N, 118.77° E) from April to June 2019. The experimental soil was silty loam (31.41% sand, 53.8% silt, and 14.8% clay) with a bulk density of 1.35 g cm⁻³. A self-designed NPI device was used, consisting of four components: a water supply tank, a negative pressure generating unit, an electronic control unit, and a ceramic emitter. This system enabled automatic pressure regulation^[22].

The experimental design included three negative pressure irrigation treatments: -2 kPa (N1), -4 kPa (N2), and -6 kPa (N3). Surface irrigation was used as the conventional surface irrigation (CK). For NPI treatments, ceramic emitters (40 cm in length) were installed in soil boxes (80 cm × 40 cm × 60 cm) at a depth of 15 cm below the soil surface, connected to the water chamber via water pipes. Water was continuously supplied at a low flow rate through the NPI system. Chinese chives were transplanted with 10 cm row spacing and 5 cm plant spacing on both sides of each emitter. For CK, the soil box dimensions were 40 cm × 40 cm × 60 cm. The irrigation volume in CK was determined based on pre-irrigation soil moisture and the field capacity (FC).

System boundary

The system boundary considered both GHG emissions and Nr losses, and was divided into two stages [Figure 1]. The first stage included upstream processes such as the production and transportation of agricultural inputs (fertilizers, irrigation equipment, labor, and electricity). The second stage covered field processes during Chinese chive cultivation, including soil organic carbon (SOC) sequestration, N₂O emissions, NO_x emissions, nitrogen leaching, nitrogen runoff, and NH₃ volatilization. The CF accounted for GHG emissions from upstream processes, SOC sequestration, and N₂O emissions. The NF included Nr losses from upstream processes as well as field losses from N₂O and NO_x emissions, nitrogen leaching, runoff, and NH₃ volatilization.

Calculations of carbon and nitrogen footprints

Carbon footprint

GHG emissions during Chinese chive production under NPI include both upstream processes (production and transportation of inputs) and field application processes. The CF was calculated as follows^[23]:

$$CF = \sum_{i=1}^n (AI_i \times EF_i) + N_2O_{total} - \Delta SOC \quad (1)$$

$$N_2O_{total} = N_2O_{direct} + (F_{N-input} \times FRAC_{gasf} \times EF_{gasf} + F_{N-input} \times FRAC_{leach} \times EF_{leach}) \times 44/28 \quad (2)$$

$$\Delta SOC = 44/12 \times NECB \times 0.213 \quad (3)$$

$$CF_y = CF/yield \quad (4)$$

where AI_i represents agricultural production input type i , and EF_i is the corresponding CO₂ emission coefficient [Supplementary Table 1]. N_2O_{total} denote total N₂O emissions (kg CO₂-eq ha⁻¹), and N_2O_{direct} is direct cumulative N₂O emission (kg CO₂-eq ha⁻¹). $F_{N-input}$ the applied nitrogen fertilizer (kg ha⁻¹). $FRAC_{GASF}$ and $FRAC_{LEACH}$ represent the fractions of inorganic N fertilizer lost as NH₃-N/NO_x-N volatilization and nitrate leaching, respectively (0.1 kg NH₃-N + NO_x-N volatilized kg⁻¹ N input, and 0.3 kg N leached kg⁻¹ N input)^[24]. EF_{gasf} and EF_{leach} are the emission factors for indirect N₂O emissions from volatilization and

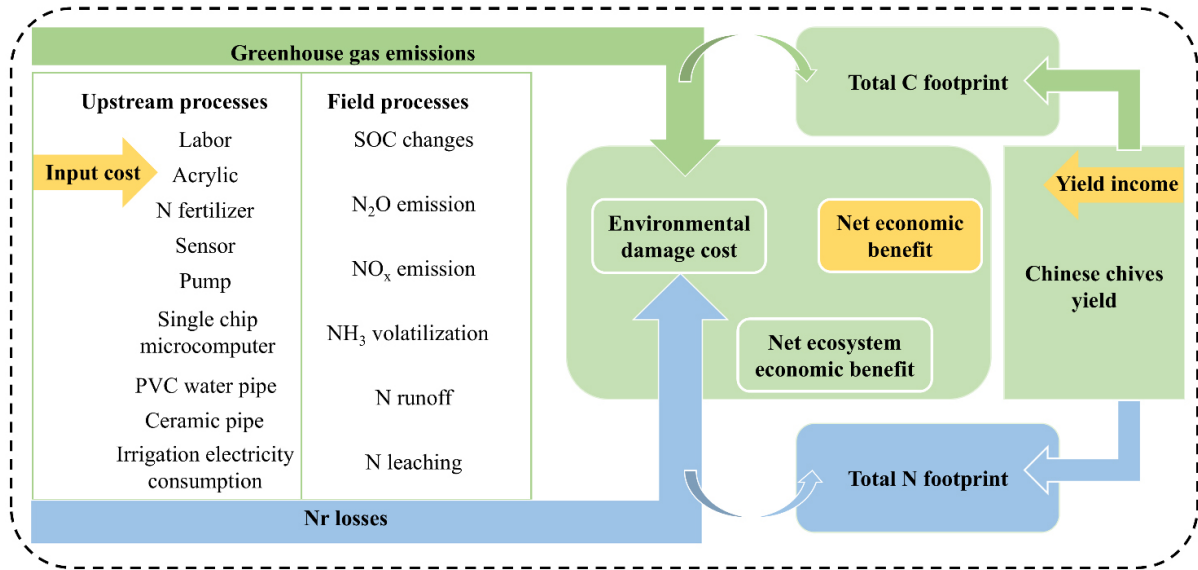


Figure 1. System boundaries for the life cycle assessment of the carbon footprint and nitrogen footprint in Chinese chive production.

leaching ($0.01 \text{ kg N}_2\text{O-N kg}^{-1} \text{ NH}_3\text{-N} + \text{NO}_x\text{-N}$ volatilized, and $0.0075 \text{ kg N}_2\text{O-N kg}^{-1} \text{ N}$ leached)^[24]. The factor $44/28$ is the molecular weight ratio of N_2O to N . NECB refers to the net ecosystem carbon budget (details in [Supplementary Text 1](#)). CF_y represents the CF per unit yield.

Nitrogen footprint

NF was assessed by quantifying nitrogen losses from field inputs, including N_2O emissions, NH_3 volatilization, and nitrogen leaching. NF (kg N-eq kg^{-1}) was calculated as follows^[25]:

$$NF_{total} = NF_{input} + NE_{\text{N}_2\text{O}} + NV_{\text{NH}_3} + NL_{\text{NO}_3^-} + NL_{\text{NH}_4^+} \quad (5)$$

$$NF_{input} = \sum_{i=1}^n AI_i \times NF_i \quad (6)$$

$$NV_{\text{NH}_3} = F_{\text{N-input}} \times \varphi \times \frac{17}{14} \times 0.833 \quad (7)$$

$$NL_{\text{NO}_3^-} = F_{\text{N-input}} \times \sigma \times \frac{64}{14} \times 0.238 \quad (8)$$

$$NL_{\text{NH}_4^+} = F_{\text{N-input}} \times \gamma \times \frac{18}{14} \times 0.786 \quad (9)$$

$$NF_y = NF_{total}/\text{yield} \quad (10)$$

where NF_{input} represents nitrogen from farm inputs, AI_i and NF_i denote agricultural input type i and its emission factor (kg N-eq ha^{-1}), respectively [[Supplementary Table 1](#)]. $NE_{\text{N}_2\text{O}}$, NV_{NH_3} , $NL_{\text{NO}_3^-}$, $NL_{\text{NH}_4^+}$, are the cumulative direct N_2O emissions, NH_3 volatilization, NH_4^+ leaching, and NO_3^- leaching from fertilizer application and straw incorporation. The coefficients are: $\varphi = \text{NH}_3$ volatilization loss coefficient (0.338)^[26]; $\sigma = \text{NO}_3^-$ leaching coefficient (0.26); $\gamma = \text{NH}_4^+$ leaching coefficient (0.85)^[27]. The factors 0.833 , 0.238 , and 0.786 represent the eutrophication potentials of NH_3 , NO_3^- , and NH_4^+ , respectively (kg N-eq kg^{-1})^[28]. NF_y represents NF per unit yield.

Calculation of net economic benefit, environmental damage cost, and NEEB

The Net Economic Benefit (NEB), (CNY ha⁻¹ yr⁻¹) was calculated as follows:

$$NEB = Revenue_{yield} - Cost_{agricultural\ input} \quad (11)$$

where $Revenue_{yield}$ represents the income from leek production [Supplementary Table 2]. $Cost_{agricultural\ input}$ is the sum of expenditures on agricultural materials, energy, and labor for vegetable cultivation, as summarized in Supplementary Table 3. All infrastructure components were assigned a specific service life based on field practice or manufacturer specifications: 15 years for ceramic emitters, 2 years for soft Polyvinyl chloride (PVC) tubing, and 5 years for control units. The costs and associated environmental burdens of equipment (e.g., the carbon and nitrogen footprints of ceramic emitters) were amortized over their respective service lives to reflect contributions across multiple growing seasons. A straight-line depreciation method was used to annualize both costs and environmental impacts. Given the prototype nature of the system, no reuse potential was assumed for any component. Detailed calculations and amortization assumptions are provided in Supplementary Text 2.

The Environmental Damage Cost (EDC) (CNY ha⁻¹ yr⁻¹), the sum of climate warming impacts from GHG emissions, soil acidification from NH₃ and NO_x emissions, and eutrophication from nitrogen leaching/runoff, was calculated as follows:

$$EDC = \sum_{i=1}^k (Nr_i \times U_{Nr,i}) + GHG_{CO_2-eq} \times U_{CO_2-eq} \quad (12)$$

where k is the total number of Nr loss types, and i denotes a specific type of Nr loss. Nr_i (kg N) is the amount of nitrogen lost in form i , and $U_{Nr,i}$ is the corresponding damage cost to human health and ecosystems per unit of nitrogen lost. The damage costs of NH₃, NO_x, and nitrogen leaching/runoff were 37.5, 29.6, and 9.3 CNY kg⁻¹ N, respectively^[29]. GHG_{CO_2-eq} is the total greenhouse gas emission (ton CO_{2-eq}), and U_{CO_2-eq} is the unit damage cost of global warming, set at 174.3 CNY ton⁻¹ CO_{2-eq}^[29].

The NEEB (CNY ha⁻¹ yr⁻¹) was then calculated as:

$$NEEB = NEB - EDC \quad (13)$$

RESULTS

Carbon and nitrogen footprints

The CF and NF varied significantly among treatments [Figure 2]. The CK treatment exhibited the highest CF value (4,519.60 kg CO₂ ha⁻¹), while all NPI treatments (N1, N2, N3) resulted in markedly lower CF values (1,148.21~1,383.37 kg CO₂ ha⁻¹). In the CK treatment, CF was primarily driven by N₂O_{total} emissions and agricultural inputs, accounting for 62.36% and 43.34%, respectively, while ΔSOC contributed only 5.12%. In contrast, CF under NPI treatments was dominated by agricultural inputs, contributing over 95%, whereas N₂O_{total} emissions accounted for only 2.57%~7.36%. The substantial CF reduction in NPI treatments was mainly attributed to significantly lower N₂O_{total} emissions compared with CK (by 95.97%~98.83%). Within agricultural inputs, N fertilizer emissions constituted 99.63% of CF in the CK treatment, while labor contributed only 0.37%. For NPI treatments, the main contributors were N fertilizer (42.78%~50.95%) and ceramic pipes (45.34%~52.90%). Other components, including labor, acrylic materials, sensors, pumps, single-chip microcontrollers, PVC water pipes, and electricity for irrigation, contributed less than 5% and were relatively minor [Supplementary Table 1 and Supplementary Table 3].

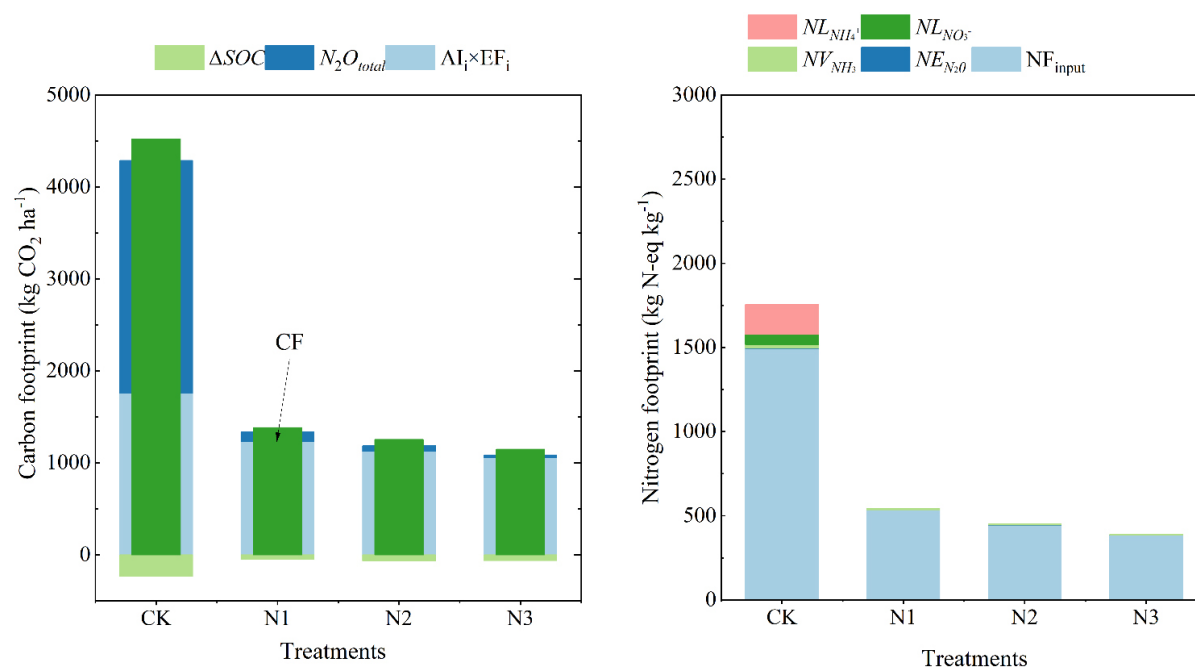


Figure 2. Changes in carbon footprints and nitrogen footprints during Chinese chives cultivation.

The inclusion of ΔSOC indicated net carbon losses across all treatments, with CK showing the greatest SOC depletion ($-231.21 \text{ kg C ha}^{-1}$). Although ΔSOC remained negative under N1, N2, and N3 (-48.91 , -65.38 , and $-62.21 \text{ kg C ha}^{-1}$, respectively), these losses were much smaller than those observed in CK. This suggests that while NPI did not achieve net SOC sequestration, it substantially reduced soil carbon loss compared with CK.

NF showed a similar trend. CK had the highest NF, with 75.99% attributed to N fertilizer input. In contrast, N1, N2, and N3 had much lower NF values, 67.76%~76.45% lower than CK. Under NPI treatments, NF primarily originated from N fertilizer emissions (93.15%~94.55%), with ceramic pipes contributing 3.95%~5.41%. As shown in Table 1, the reduced NF in NPI treatments was largely due to lower N fertilizer application rates, with N emissions from fertilizer 64.15%~74.20% lower than in CK. Among NPI treatments, N1 had the highest NF, followed by N2 and N3, suggesting a potential trade-off between irrigation innovation and reactive nitrogen losses.

NEB

The $\text{Cost}_{\text{agricultural input}}$ was shown in Table 2. The $\text{Cost}_{\text{agricultural input}}$ under NPI treatments was 0.99–2.01 times higher than that in CK. Compared with CK, labor costs in the N1, N2, and N3 treatments were reduced by 89.33%, and nitrogen fertilizer costs were reduced by 64.15%~74.20%. However, the use of negative-pressure irrigation equipment incurred additional expenditures for components such as acrylic materials, sensors, pumps, single-chip microcomputers, and PVC pipes. These items accounted for 1.35%~1.37% of $\text{Cost}_{\text{agricultural input}}$. The most significant cost under NPI was for ceramic pipes, which amounted to 2,078.90 CNY ha⁻¹ and represented 88.36%~89.80% of $\text{Cost}_{\text{agricultural input}}$. In contrast, in CK, the largest cost was labor, accounting for 67.90% of total inputs.

The $\text{Revenue}_{\text{yield}}$ for CK, N1, N2, and N3 was 2,950.46, 2,905.59, 2,391.23, and 2,097.04 CNY ha⁻¹, respectively. There was no significant difference in $\text{Revenue}_{\text{yield}}$ between N1 and CK. However, $\text{Revenue}_{\text{yield}}$

Table 1. Emission inventory by component of agricultural inputs

Items	Emissions from agricultural inputs (kg CO ₂ ha ⁻¹)				NF _{input} (kg N-eq ha ⁻¹)			
	CK	N1	N2	N3	CK	N1	N2	N3
Labor	6.45	0.69	0.69	0.69	/	/	/	/
N fertilizer	1,751.83	628.08	521.27	451.97	1,492.92	535.25	444.23	385.17
NPI control unit	8.44	8.44	8.44	8.44	/	/	/	/
Acrylic	0.00	0.002	0.002	0.002	/	/	/	/
Air pressure sensor	0.00	0.002	0.002	0.002	/	/	/	/
Liquid level sensor	0.02	0.02	0.02	0.02	/	/	/	/
Vacuum pump	0.02	0.02	0.02	0.02	/	/	/	/
Single-chip microcomputer	0.02	0.02	0.02	0.02	/	/	/	/
Peristaltic pump	0.02	0.02	0.02	0.02	/	/	/	/
PVC water pipe	0.00	1.02	1.02	1.02	/	/	/	/
Ceramic pipe	0.00	558.91	558.91	558.91	0.00	22.36	22.36	22.36
Irrigation electricity consumption	0.00	35.42	35.42	35.42	0.00	0.00144	0.00144	0.00144

CK: Conventional surface irrigation; PVC: polyvinyl chloride.

Table 2. Agricultural Input Inventory and total cost per treatment (CNY ha⁻¹)

Item	Cost _{agriculturalinput}			
	CK	N1	N2	N3
Labor	795	84.80	84.80	84.80
N fertilizer	375.84	134.75	111.83	96.99
NPI control unit	0	25.32	25.32	25.32
PVC water pipe	0	6.47	6.47	6.47
Ceramic pipe	0	2,078.90	2,078.90	2,078.90
Irrigation electricity consumption	0	22.46	22.46	22.46
Total	1,170.84	2,352.71	2,329.80	2,314.93

Note: "Total" refers to the sum of all listed input costs under each treatment, including labor, fertilizer, irrigation equipment, and electricity consumption. NPI: Negative-pressure irrigation; CK: conventional surface irrigation; PVC: polyvinyl chloride.

under N2 and N3 were 18.95% and 28.92% lower than CK, respectively. The NEB was highest in CK at 1,779 CNY ha⁻¹, while all NPI treatments produced lower NEB values. Notably, the NEB under N3 was negative [Figure 3].

EDC and NEEB

As shown in Figure 4, the EDC under the CK treatment were significantly higher than those under NPI. Specifically, the EDC values for N1, N2, and N3 were 558, 479, and 423 CNY ha⁻¹, respectively, whereas CK reached approximately 3,908 CNY ha⁻¹. This reduction in EDC under NPI was attributed to its improved environmental performance through lower reactive nitrogen emissions and a reduced carbon footprint.

All treatments exhibited substantially negative NEEB. In the NPI treatments, although the NEB was significantly lower than that of CK, the EDC was also markedly reduced. However, this reduction in EDC was insufficient to offset the decline in NEB, resulting in significantly lower NEEB values compared with CK. In contrast, the N1 treatment yielded a near-zero NEEB (-10 CNY ha⁻¹), suggesting a relatively better ecological-economic balance under conventional management.

DISCUSSION

Upstream contributions to carbon and nitrogen footprints

When accounting for the embedded emissions of NPI equipment, NPI still exhibited lower CF, NF, CF_y,

and NF_y compared with conventional irrigation [Figure 2 and Supplementary Figure 1]. The reduction in CF under NPI was primarily driven by a significant decrease in N_2O_{total} emissions, which resulted from both reduced N_2O_{direct} emissions under NPI and lower N fertilizer application rates, which in turn led to decreased NH_3 -N and NO_x -N emissions. Although the adoption of NPI increased emissions from agricultural inputs due to additional equipment requirements, this increase was outweighed by the larger reductions associated with decreased N fertilizer use. Consequently, input-related emissions were ultimately lower under NPI, contributing to a significantly reduced CF and, in turn, a lower CF_y . The NF under NPI was also significantly lower than under CK. This reduction was mainly due to the decreased N fertilizer application rate, with fertilizer-related N emissions in the NPI treatment being 64.15%-74.20% lower than in the CK treatment.

We initially assumed a service lifespan of 15 years for the ceramic pipes. However, under actual soil environments, porous ceramics are prone to clogging and functional failure, which could bias estimates of carbon and nitrogen emissions. Because empirical data and industry standards on service life are lacking, we also examined shorter assumed lifespans of 10 and 5 years. The results showed that at a 10-year lifespan, CFs were 1,694.35, 1,558.46, and 1,450.35 kg CO_2 -eq ha^{-1} , and NFs were 577.31, 484.75, and 424.68 kg N-eq ha^{-1} , respectively-representing increases of 19.75%-23.87% in CF and 1.97%-2.70% in NF relative to a 15-year lifespan [Figure 5]. At a 5-year lifespan, CFs rose to 2,532.72, 2,396.83, and 2,288.72 kg CO_2 -eq ha^{-1} , and NFs to 610.84, 518.29, and 458.21 kg N-eq ha^{-1} -corresponding to increases of 44.14%-48.84% in CF and 7.81%-10.02% in NF compared with the 15-year lifespan. These findings suggest that ceramic pipe lifespan has a greater impact on CF than on NF, likely due to the higher emission factor for CO_2 compared with reactive nitrogen species. In this study, the emission factor for ceramic pipes was assumed to be 2.5 kg CO_2 -eq kg^{-1} -a conservative estimate derived from the literature [Supplementary Table 1]-due to the absence of specific LCA data for porous ceramic irrigation emitters. This introduces uncertainty into the upstream CF estimates and may underestimate the actual environmental burden.

These results highlight a well-documented concern in LCA: advanced technologies that improve field-level efficiency can simultaneously introduce substantial upstream environmental burdens^[30,31]. Specifically, the use of high-temperature sintered ceramics and PVC infrastructure entails considerable embodied carbon, which can offset reductions achieved in field emissions. This trade-off underscores the importance of integrated assessment frameworks that move beyond operational performance metrics (e.g., water or fertilizer use efficiency) to fully capture the cradle-to-grave environmental costs of agricultural innovations. To enhance the sustainability of NPI systems, future efforts should focus on optimizing material choices (e.g., adopting low-carbon or recycled inputs), extending the lifespan and reusability of emitters, and exploring alternative emitter designs such as biodegradable polymers or composite materials. Incorporating these scenarios into future LCA models will be critical for guiding sustainable engineering practices and informing relevant policy frameworks.

From cost burden to environmental compensation: reframing NPI viability

The NEB analysis showed that although the NPI system significantly reduced operational costs - particularly labor (by ~89%) and nitrogen fertilizer inputs (by 64%~75%) - the high cost of system components, especially the ceramic irrigation devices, led to a substantial increase in total investment. Ceramic components alone accounted for approximately 88%~90% of total costs (2,078.90 CNY ha^{-1}), far outweighing the savings from reduced fertilizer and labor. This finding aligns with previous studies that identified high initial capital requirements as a primary barrier to adopting precision or novel irrigation systems in smallholder and resource-limited settings^[31-33].

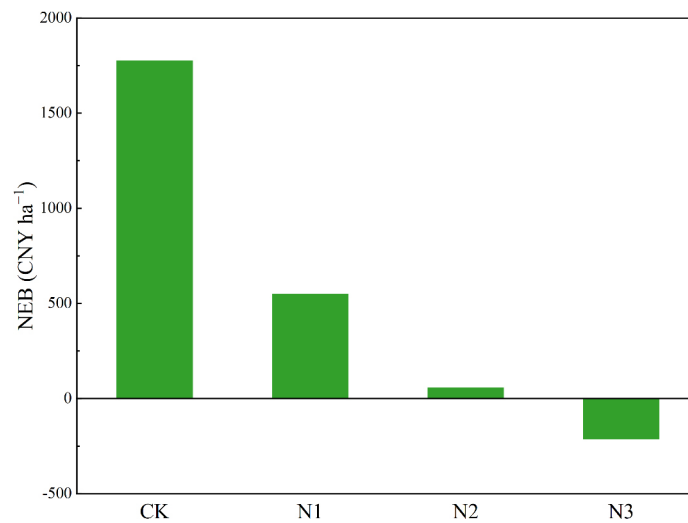


Figure 3. Changes in Net Economic Benefit (NEB) during Chinese chives cultivation.

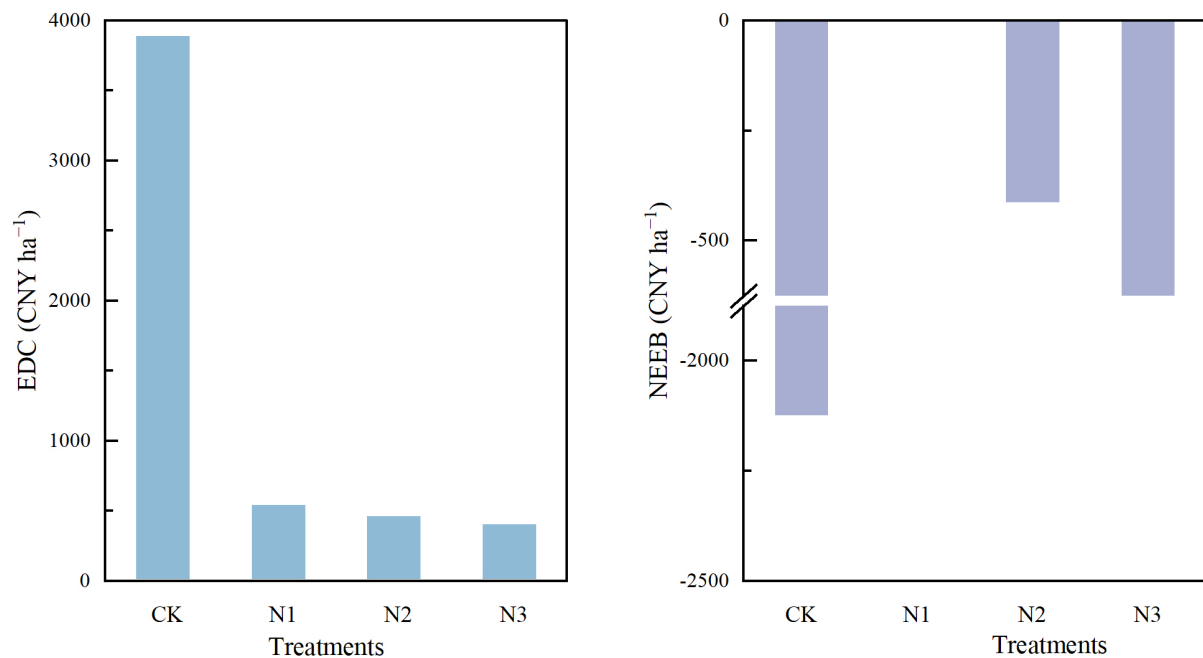


Figure 4. Changes in environmental damage costs (EDC) and net ecosystem economic benefit (NEEB) during Chinese chives cultivation.

Crop yield revenue did not significantly increase under NPI, and in some cases, even declined slightly under treatments N2 and N3. The slight yield reduction observed under these treatments may be attributed to suboptimal soil moisture conditions. Specifically, excessively low soil water potential likely restricted crop water uptake and growth, thereby reducing yield performance^[15]. Due to the mismatch between input costs and revenue, the NEB under all NPI treatments was lower than under the CK treatment, turning negative under the N3 treatment. This indicates that, although the system is agronomically efficient (e.g., in water and fertilizer savings), its short-term economic feasibility is limited unless substantial financial support mechanisms -such as subsidies or carbon market incentives- are introduced to offset the cost gap.

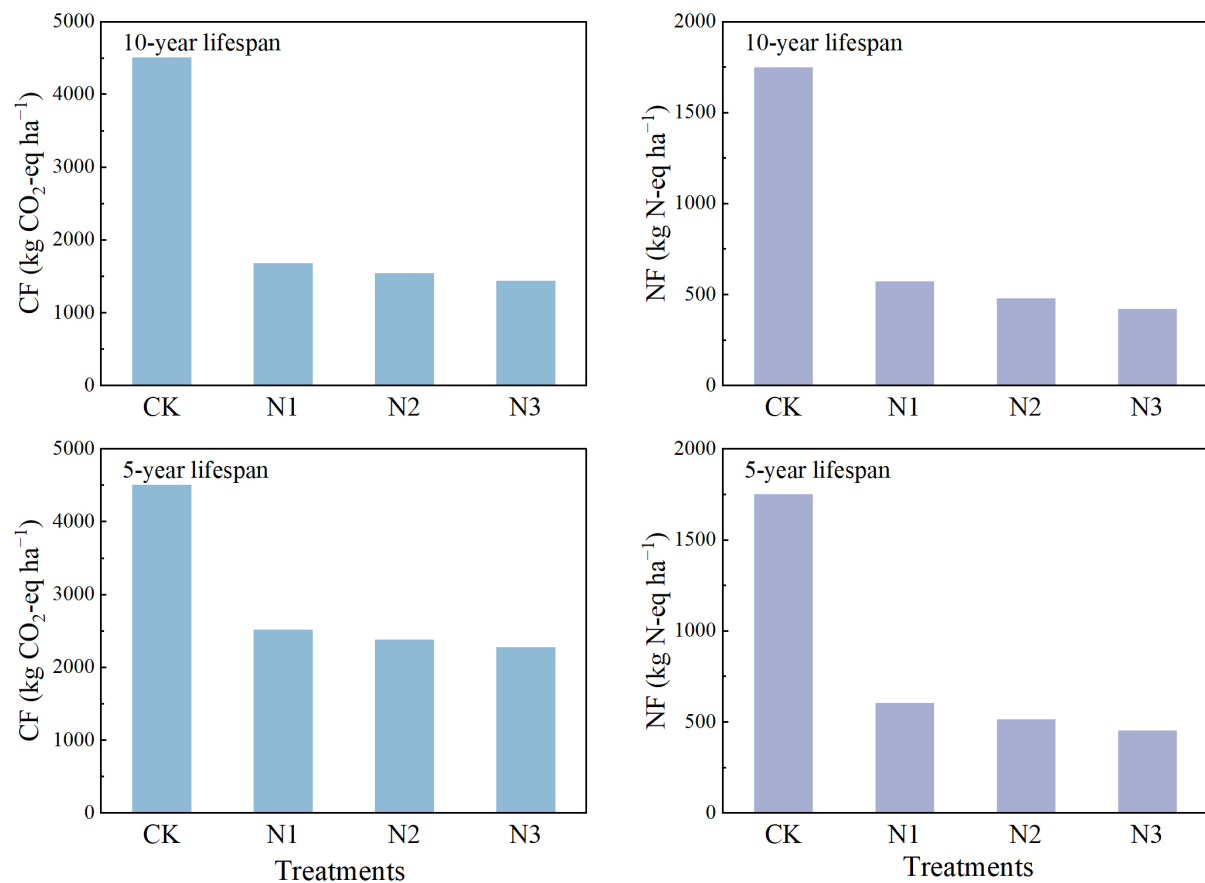


Figure 5. Carbon and nitrogen footprints under different service lifespans of ceramic emitters.

Although the NEEB under the NPI system remained negative, it was consistently higher than that of the CK treatment. This suggests that despite the lower NEB resulting from high equipment costs and limited yield improvements, the NPI system provided notable environmental advantages. Specifically, EDC values were significantly lower under NPI, primarily due to marked reductions in greenhouse gas emissions and nitrogen losses. Since NEEB integrates both economic performance and environmental externalities, the relatively higher NEEB under NPI indicates a more favorable overall sustainability profile. In other words, while NPI may not currently offer short-term economic returns, its environmental co-benefits can partially offset economic drawbacks, highlighting its potential as a more sustainable irrigation strategy in the long term—particularly if policy instruments such as subsidies or carbon credits are introduced to internalize these environmental gains. It should be noted, however, that the EDC estimation relied on average unit damage costs for pollutants, without adjustments for regional characteristics (e.g., ecological sensitivity or population density). While this approach enables meaningful relative comparisons, it introduces uncertainty into the absolute values. Future assessments should incorporate region-specific damage cost parameters to enhance accuracy and policy relevance.

This finding aligns with results from integrated assessment models, which show that even technologies with high ecosystem service value may face adoption barriers without appropriate policy support^[34]. In this study, the monetized value of environmental benefits (e.g., reduced emissions and pollution) failed to offset the substantial investment costs. This underscores the need for mechanisms such as carbon credit markets, ecological compensation, or green subsidies to internalize external environmental gains.

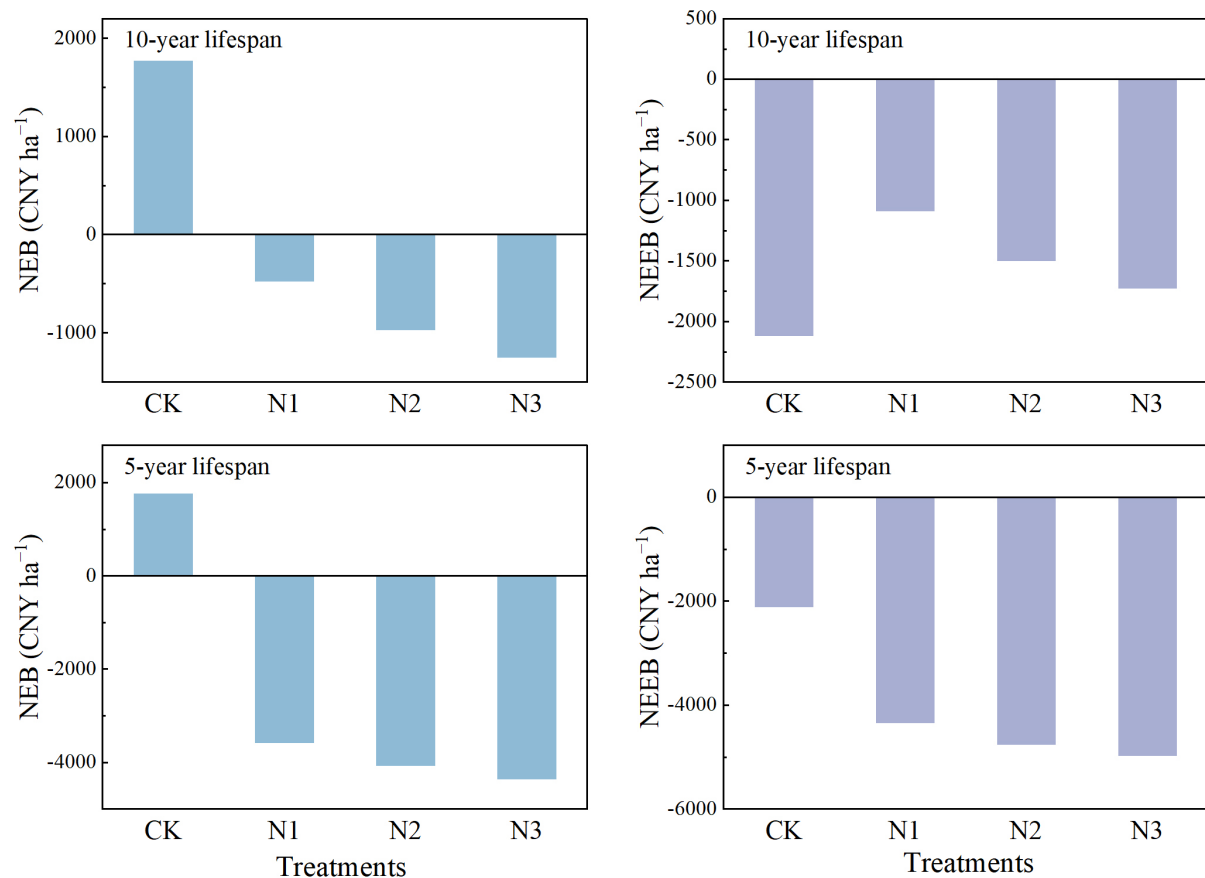


Figure 6. Changes in Net Economic Benefit (NEB) and net ecosystem economic benefit (NEEB) under different service lifespans of ceramic emitters.

In this study, a 15-year lifespan for ceramic emitters was assumed based on design specifications. However, in real soil environments, issues such as clogging may shorten their actual service life, thereby affecting cost allocation calculations. When the lifespan was assumed to be 10 years, the NEB under NPI treatments remained negative, ranging from -485.57 CNY ha⁻¹ to -1,257.34 CNY ha⁻¹, which were 127.34% to 170.65% lower than those of the CK treatment. Under the same conditions, NEEB ranged from -1,733.14 CNY ha⁻¹ to -1,098.53 CNY ha⁻¹, representing an 18.57% to 48.39% improvement over CK [Figure 6]. However, assuming a lifespan of only 5 years, NEB values dropped sharply to -4,375.69 to -3,604.93 CNY ha⁻¹, or 2.03 to 2.46 times lower than CK, while NEEB ranged from -4,997.63 to -4,363.01 CNY ha⁻¹, or 1.05 to 1.35 times lower than CK [Figure 6]. These results demonstrate that the assumed service life of ceramic emitters significantly influences evaluations of both NEB and NEEB. Therefore, future research involving accelerated aging tests or in-situ durability assessments would be valuable for more accurately estimating their actual lifespan and corresponding economic performance. Additionally, this life cycle assessment did not account for emissions associated with equipment transportation and disposal. Although these stages may contribute some environmental impacts, the dominance of manufacturing emissions suggests their omission does not significantly alter the overall environmental assessment.

This study demonstrates that NPI systems impose a considerable upstream environmental burden, primarily due to ceramic emitters, which dominate the system's carbon footprint. By contrast, conventional irrigation technologies-such as plastic drip irrigation-typically have lower embodied carbon due to material

and manufacturing differences, though they may pose challenges related to durability and recyclability. Although a detailed life cycle comparison was beyond the scope of this study, future research should investigate the environmental and economic trade-offs among different irrigation technologies to inform more sustainable infrastructure choices. Furthermore, NPI systems may provide additional long-term ecological benefits, such as improved soil structure, reduced salinity, and enhanced microbial resilience. However, these potential advantages were not captured in this short-term analysis due to its limited duration. Future research should incorporate such long-term ecological contributions into life cycle assessments to enable a more comprehensive evaluation of the environmental-economic synergies and sustainability potential of NPI systems.

CONCLUSION

This study applied an LCA framework to comprehensively evaluate the environmental and economic performance of a NPI system in protected agriculture. Compared with conventional irrigation, NPI reduced CF and NF by up to 75% and 76%, respectively, primarily through reduced N₂O emissions and fertilizer inputs. It also mitigated soil carbon losses, demonstrating strong potential to enhance environmental sustainability. However, the high upstream cost and emissions associated with ceramic emitters (accounting for nearly 90% of input costs) resulted in lower NEB under NPI treatments, with some scenarios even producing negative returns. While all treatments showed negative NEEB, NPI treatments-especially N1-achieved significantly higher NEEB than CK, with N1 approaching break-even. This indicates improved ecological-economic efficiency despite reduced financial returns and highlights NPI's potential to support sustainable agricultural development under optimized conditions. To promote broader adoption of NPI systems, future efforts should prioritize enhancing the manufacturing efficiency and durability of emitters, developing low-carbon or renewable materials (e.g., bio-based ceramics), and integrating environmental policy tools such as carbon trading or ecological subsidies. Additionally, comparative life cycle analyses with other water-saving technologies (e.g., drip and micro-sprinkler irrigation) are essential to assess trade-offs and guide sustainable irrigation strategies across diverse agricultural contexts.

DECLARATIONS

Author's contributions

Conceptualization, formal analysis, funding acquisition, investigation, methodology, writing - original draft: Wang, K.

Conceptualization, methodology, writing - review and editing: Chen, P.

Formal analysis, investigation: Min, Z.

Investigation, writing - review and editing: Chen, S.

Validation, writing - review and editing: Zhou, Z.

Availability of data and materials

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Conflicts of interest

Min, Z. is an employee of Powerchina Zhongnan Engineering Corporation Limited and contributed to this work through formal analysis and investigation. The other authors declared that there are no conflicts of interest.

Ethical approval and consent to participate

Not applicable

Consent for publication

Not applicable.

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