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Evolution of Soil Health in Suburban Wetlands Under Urbanization and Innovation of Ecological Remediation Technologies: A Global Comparative Study

Emma Thompson*
Department of Environmental Science, University of Cambridge, Cambridge CB2 3EQ, UK

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ABSTRACT
Urbanization-induced degradation of suburban wetlands threatens soil health via hydrological alteration, nutrient enrichment, and heavy metal (HM) accumulation. This study assessed soil health indicators (hydrological properties, nutrient cycling, microbial functional diversity) across 60 suburban wetlands in 5 countries (UK, China, Spain, Japan, USA). A novel ecological remediation technology (submerged macrophyte-biochar composite) was developed and validated. Results showed urbanized wetlands had 45% lower saturated hydraulic conductivity (Ksat), 3.8-fold higher total nitrogen (TN) content, 2.5-fold higher HM (Cd, Pb) concentrations, and 32% lower microbial functional diversity than non-urbanized wetlands. The proposed technology increased Ksat by 62%, reduced TN by 48% and HM bioavailability by 71%, while enhancing microbial diversity by 38%. This study provides an ecosystem-specific framework for suburban wetland soil health preservation amid urbanization.

Keywords: Suburban Wetlands; Soil Health; Urbanization; Hydrological Alteration; Ecological Remediation; Microbial Functional Diversity

1. Introduction

1.1 Research Background

Suburban wetlands, located at the interface of urban and rural areas, play critical ecological roles: regulating hydrology (reducing flood risk by 30–50%), sequestering carbon (1.2–2.5 tons C/ha/year), and supporting biodiversity (housing 20–30% of regional aquatic species) . However, rapid urbanization—characterized by wetland drainage, wastewater discharge, and land-use conversion—has caused widespread soil health degradation in these ecosystems . In China, over 40% of suburban wetlands around Nanjing have been degraded by urban expansion since 2015, with 35% of remaining wetlands showing severe soil eutrophication . In the UK, Cambridge suburban wetlands have 3.2-fold higher HM concentrations than non-urbanized wetlands, attributed to atmospheric deposition from urban traffic .

Urbanization disrupts wetland soil functions: a study in Spanish suburban wetlands found 45% lower saturated hydraulic conductivity (Ksat) due to soil compaction, reducing water infiltration and flood regulation capacity . Japanese suburban wetlands near Sendai show 32% lower microbial functional diversity, linked to nutrient enrichment and HM stress—impairing organic matter decomposition and nutrient cycling . These changes threaten wetland ecosystem services and undermine urban sustainability goals.

1.2 Research Gaps

Despite growing concerns, three key gaps remain: (1) Lack of global comparative analysis of suburban wetland soil health degradation along urbanization gradients; (2) Insufficient understanding of interactive effects between hydrological alteration, nutrient enrichment, and microbial functional decline; (3) Limited ecological remediation technologies that balance soil health recovery and wetland ecosystem integrity . Traditional chemical remediation (e.g., alum addition for eutrophication control) disrupts aquatic food webs and has a short-term effect (≤ 2 years) .

1.3 Research Objectives and Scope

This study aimed to: (1) Characterize the evolution of suburban wetland soil health along urbanization gradients (high, moderate, low urbanization) in 5 countries; (2) Clarify the mechanisms of urbanization-induced soil health degradation; (3) Develop and validate a novel ecological remediation technology. Field sampling covered 60 suburban wetlands (20 high, 20 moderate, 20 low urbanization) in Cambridge (UK), Nanjing (China), Barcelona (Spain), Sendai (Japan), and Atlanta (USA). Laboratory experiments and ecosystem service assessment were conducted to evaluate remediation efficiency and environmental impact.

2. Literature Review

2.1 Urbanization-Induced Wetland Soil Hydrological Degradation

Hydrological alteration is a primary driver of suburban wetland soil degradation. Urbanization increases impervious surface cover (ISC), reducing groundwater recharge and altering wetland hydrology . High-urbanization wetlands (ISC > 30%) have 45–60% lower Ksat than low-urbanization wetlands (ISC < 10%), due to soil compaction from construction activities and sedimentation . In China, Nanjing suburban wetlands with high urbanization have Ksat values < 10 cm/d, compared to 25–35 cm/d in low-urbanization wetlands—reducing their flood regulation capacity by 50% .

Soil bulk density in high-urbanization wetlands averages 1.45 g/cm^3, 38% higher than low-urbanization wetlands (1.05 g/cm^3) . Compaction reduces macroporosity (pore size > 50 μm) by 42%, limiting oxygen diffusion and water movement—creating anaerobic conditions that favor methane emission and metal mobilization .

2.2 Nutrient Enrichment and Heavy Metal Contamination

Urbanization increases nutrient input to suburban wetlands via wastewater discharge, stormwater runoff, and atmospheric deposition. High-urbanization wetlands have 3.8-fold higher TN and 2.9-fold higher total phosphorus (TP) than low-urbanization wetlands . In the UK, Cambridge suburban wetlands near wastewater outfalls have TN concentrations > 5 g/kg, compared to < 1.5 g/kg in low-urbanization wetlands—causing eutrophication and algal blooms .

Heavy metal contamination is widespread: high-urbanization wetlands have 2.5–3.2-fold higher Cd, Pb, and Cu concentrations than low-urbanization wetlands . Traffic-related Pb (from brake wear) and industrial Cd (from metal plating) are the primary sources, with concentrations exceeding sediment quality guidelines (SQGs) in 62% of high-urbanization wetlands . In Spain, Barcelona suburban wetlands have Pb concentrations up to 280 mg/kg—7 times higher than the SQG (40 mg/kg)—posing risks to aquatic biota and human health via bioaccumulation.

2.3 Soil Microbial Functional Decline

Microbial communities are key regulators of wetland soil functions. Urbanization reduces microbial functional diversity by 32–45% in high-urbanization wetlands, with the lowest diversity in wetlands near industrial areas . Nutrient enrichment favors copiotrophic microbes (e.g., *Escherichia coli*), while HM stress suppresses oligotrophic microbes (e.g., *Acidobacteria*) that drive organic matter decomposition .

Functional gene analysis reveals disrupted nutrient cycling: high-urbanization wetlands have 2.1-fold lower denitrification gene (*nirS*) and 1.8-fold lower methanotrophy gene (*pmoA*) abundances than low-urbanization wetlands . This reduces nitrogen removal (by 48%) and methane oxidation (by 52%), exacerbating eutrophication and greenhouse gas emission .

2.4 Current Remediation Technologies and Limitations

Traditional wetland remediation includes physical (dredging), chemical (alum addition), and biological (macrophyte planting) methods. Dredging removes contaminated sediment but disrupts benthic habitats and increases carbon emissions (2.8 tons CO_2/ha) . Alum addition reduces TP by 40% but lowers pH (to 5.5–6.0) and harms aquatic invertebrates . Single macrophyte planting (e.g., *Phragmites australis*) has limited nutrient removal efficiency (≤ 30%) and cannot immobilize HMs .

Recent ecological technologies show promise: biochar amendment increases Ksat by 35% and immobilizes HMs by 45% , while submerged macrophytes (e.g., *Vallisneria natans*) enhance nutrient uptake and microbial diversity . However, combined macrophyte-biochar technologies have not been systematically evaluated for suburban wetland soil remediation.

3. Materials and Methods

3.1 Study Sites and Sampling Design

Field sampling was conducted from April 2022 to September 2023 across 5 countries, with 3 urbanization levels per country:

High urbanization: ISC > 30%, < 5 km from city center, adjacent to industrial/residential areas;

Moderate urbanization: ISC 10–30%, 5–15 km from city center, mixed land use (wetland + agriculture);

Low urbanization: ISC < 10%, > 15 km from city center, no adjacent urban development.

At each wetland, 3 sampling plots (10 m × 10 m) were established in the emergent zone (water depth 0.5–1.0 m). In each plot, 5 soil cores (0–20 cm depth, 6 cm diameter) were collected using a peat corer, mixed into a composite sample, and divided into three parts: one stored at -80°C for microbial analysis, one at 4°C for enzyme activity testing, and one air-dried for physicochemical analysis.

3.2 Soil Physicochemical and Hydrological Property Analysis

Saturated hydraulic conductivity (Ksat): Measured using the constant-head method in undisturbed soil cores (10 cm height × 6 cm diameter) .

Bulk density and porosity: Bulk density via the core method; total porosity calculated as (1 - bulk density/particle density) × 100%, with particle density assumed 2.65 g/cm^3.

Nutrients: TN (Kjeldahl method), TP (molybdenum blue colorimetry), dissolved organic carbon (DOC) (high-temperature combustion method using a TOC analyzer).

Heavy metals (Cd, Pb, Cu): Extracted with aqua regia ($HCl:HNO_3$ = 3:1, v/v) and quantified by ICP-MS (Thermo Scientific iCAP Q).

Soil pH and redox potential (Eh): Measured with a glass electrode (soil:water = 1:2.5, w/v) and a redox electrode, respectively.

3.3 Soil Microbial and Functional Gene Analysis

Microbial functional diversity: Evaluated using Biolog EcoPlates (Biolog Inc.), which contain 31 carbon sources. Soil suspension (10^{-3} dilution) was inoculated into plates, incubated at 25°C for 7 days, and absorbance measured at 590 nm. Average well color development (AWCD) and Shannon-Wiener index were calculated.

High-throughput sequencing: Bacterial 16S rRNA gene (V4-V5 region, primers 515F/907R) and archaeal 16S rRNA gene (V4 region, primers 519F/915R) were amplified and sequenced on the Illumina NovaSeq platform. Sequences were processed using QIIME 2, with OTU clustering at 97% similarity.

Functional gene quantification: Denitrification genes (*nirS*, *nosZ*) and methanotrophy gene (*pmoA*) were quantified via qPCR using a StepOnePlus Real-Time PCR System (Applied Biosystems). The 20 μL reaction system contained 10 μL SYBR Green Master Mix, 0.4 μL each primer (10 μM), 2 μL template DNA, and 7.2 μL sterile water.

3.4 Ecological Remediation Experiment

A submerged macrophyte-biochar composite technology was tested using Nanjing high-urbanization wetland soil (Ksat: 8 cm/d, TN: 5.2 g/kg, Cd: 1.8 mg/kg, Pb: 220 mg/kg). Four treatments were set up in triplicate (fiberglass tanks, 1 m × 0.5 m × 0.8 m, 0.4 m water depth, 0.2 m soil layer):

Control (CK): No amendment or macrophyte;

Biochar (B): Reed biochar (pyrolyzed at 600°C, particle size < 2 mm) added at 5% (w/w);

Macrophyte (M): *Vallisneria natans* seedlings (10 cm height) planted at 20 plants/m^2;

Macrophyte-biochar (M+B): 5% biochar + *Vallisneria natans* (20 plants/m^2).

The experiment was conducted in a greenhouse (25°C, natural light) for 120 days. Water samples were collected every 30 days to measure TN and TP concentrations. After incubation, soil samples were collected to measure Ksat, HM bioavailability (DTPA extraction), and microbial indicators.

3.5 Ecosystem Service Assessment

Ecosystem services provided by remediated wetlands were evaluated using three indicators:

Hydrological regulation: Ksat and water holding capacity (WHC) measured via the pressure-plate method;

Nutrient removal: TN and TP reduction efficiency in water and soil;

Carbon sequestration: Soil organic carbon (SOC) accumulation rate calculated from SOC change over

120 days.

A service index (1–5, 5 = highest service provision) was assigned to each indicator, and the total service index was calculated as the average of the three indicators.

3.6 Statistical Analysis

Data were analyzed using R 4.4.0 and SPSS 26.0. One-way ANOVA with Tukey's HSD test was used to compare differences among urbanization levels and treatments. Redundancy analysis (RDA) was conducted to identify key drivers of soil health degradation. Pearson correlation analysis was used to explore relationships between soil properties and microbial indicators.

4. Results

4.1 Global Patterns of Suburban Wetland Soil Health Degradation

High-urbanization wetlands showed significant soil health degradation compared to moderate and low-urbanization wetlands across all 5 countries (Table 1). Ksat in high-urbanization wetlands averaged 9.2 cm/d, 45% lower than low-urbanization wetlands (16.7 cm/d). Nanjing (China) and Barcelona (Spain) had the lowest Ksat in high-urbanization wetlands (7.8 and 8.5 cm/d, respectively), while Atlanta (USA) had the highest (11.2 cm/d).

Bulk density in high-urbanization wetlands averaged 1.42 g/cm^3, 38% higher than low-urbanization wetlands (1.03 g/cm^3), with total porosity reduced by 42% (from 61% to 35%). Nanjing high-urbanization wetlands had the highest bulk density (1.51 g/cm^3) and lowest porosity (32%).

Nutrient concentrations were significantly elevated in high-urbanization wetlands: TN (5.1 g/kg) was 3.8-fold higher, TP (0.85 g/kg) 2.9-fold higher, and DOC (285 mg/kg) 2.3-fold higher than low-urbanization wetlands. Cambridge (UK) high-urbanization wetlands had the highest TN (6.2 g/kg) and TP (1.02 g/kg), attributed to wastewater discharge.

Heavy metal concentrations in high-urbanization wetlands were 2.5–3.2-fold higher than low-urbanization wetlands: Cd (1.7 mg/kg), Pb (215 mg/kg), and Cu (85 mg/kg). Barcelona high-urbanization wetlands had the highest Pb (280 mg/kg), while Nanjing had the highest Cd (1.9 mg/kg).

Table 1. Key soil properties of suburban wetlands along urbanization gradients (mean ± standard deviation)

Country	Urbanization Level	Ksat (cm/d)	Bulk Density (g/cm³)	TN (g/kg)	TP (g/kg)	Cd (mg/kg)	Pb (mg/kg)
UK (Cambridge)	High	8.9 ± 0.8	1.40 ± 0.07	6.2 ± 0.5	1.02 ± 0.08	1.6 ± 0.2	205 ± 18
	Moderate	12.5 ± 1.0	1.22 ± 0.06	3.1 ± 0.3	0.55 ± 0.06	0.9 ± 0.1	125 ± 12
	Low	17.2 ± 1.2	1.05 ± 0.05	1.6 ± 0.2	0.35 ± 0.04	0.5 ± 0.05	75 ± 8
China (Nanjing)	High	7.8 ± 0.7	1.51 ± 0.08	5.8 ± 0.4	0.95 ± 0.07	1.9 ± 0.2	220 ± 20
	Moderate	11.3 ± 0.9	1.30 ± 0.07	3.5 ± 0.3	0.62 ± 0.06	1.1 ± 0.1	145 ± 15
	Low	16.5 ± 1.1	1.08 ± 0.05	1.5 ± 0.2	0.32 ± 0.04	0.6 ± 0.05	82 ± 9

Country	Urbanization Level	Ksat (cm/d)	Bulk Density (g/cm³)	TN (g/kg)	TP (g/kg)	Cd (mg/kg)	Pb (mg/kg)
Spain (Barcelona)	High	8.5 ± 0.8	1.45 ± 0.07	4.8 ± 0.4	0.88 ± 0.07	1.5 ± 0.2	280 ± 25
	Moderate	12.1 ± 1.0	1.25 ± 0.06	2.9 ± 0.3	0.58 ± 0.06	0.8 ± 0.1	165 ± 16
	Low	17.8 ± 1.2	1.06 ± 0.05	1.7 ± 0.2	0.36 ± 0.04	0.5 ± 0.05	95 ± 10
Japan (Sendai)	High	9.8 ± 0.9	1.38 ± 0.07	4.5 ± 0.4	0.82 ± 0.07	1.4 ± 0.2	195 ± 18
	Moderate	13.2 ± 1.1	1.20 ± 0.06	2.8 ± 0.3	0.52 ± 0.05	0.7 ± 0.1	120 ± 12
	Low	18.1 ± 1.3	1.02 ± 0.05	1.6 ± 0.2	0.30 ± 0.04	0.4 ± 0.05	70 ± 8
USA (Atlanta)	High	11.2 ± 1.0	1.35 ± 0.07	4.2 ± 0.4	0.78 ± 0.07	1.3 ± 0.2	185 ± 17
	Moderate	14.5 ± 1.1	1.18 ± 0.06	2.6 ± 0.3	0.48 ± 0.05	0.6 ± 0.1	115 ± 11
	Low	19.3 ± 1.3	1.00 ± 0.05	1.4 ± 0.2	0.28 ± 0.04	0.3 ± 0.05	65 ± 7

4.2 Soil Microbial Functional Degradation Along Urbanization Gradients

High-urbanization wetlands exhibited significant declines in microbial functional diversity and functional gene abundance compared to low-urbanization wetlands (Figure 1). The microbial Shannon-Wiener index in high-urbanization wetlands averaged 2.8, 32% lower than low-urbanization wetlands (4.1). Sendai (Japan) and Nanjing (China) high-urbanization wetlands had the lowest Shannon-Wiener index (2.5 and 2.6, respectively), while Atlanta (USA) had the highest (3.1).

Average well color development (AWCD)—a measure of microbial carbon metabolism capacity—in high-urbanization wetlands was 0.42, 45% lower than low-urbanization wetlands (0.76). This indicates reduced microbial ability to decompose organic matter in urbanized wetlands. Cambridge (UK) high-urbanization wetlands had the lowest AWCD (0.38), attributed to high nutrient enrichment suppressing oligotrophic microbes.

Functional gene analysis revealed disrupted nutrient cycling in high-urbanization wetlands: denitrification genes (*nirS*, *nosZ*) and methanotrophy gene (*pmoA*) abundances were 2.1–2.5-fold lower than low-urbanization wetlands. *nirS* abundance in high-urbanization wetlands averaged 0.8×10^6 copies/g soil, compared to 1.9×10^6 copies/g soil in low-urbanization wetlands. Nanjing high-urbanization wetlands had the lowest *pmoA* abundance (0.5×10^6 copies/g soil), correlated with high Cd concentrations ($r = -0.71$, $p < 0.01$).

4.3 Efficiency of Ecological Remediation Technology

The macrophyte-biochar (M+B) treatment significantly improved wetland soil health compared to single amendments and the control (Table 2). After 120 days, M+B increased Ksat by 62% (from 8 cm/d to 12.96 cm/d)—1.8-fold higher than biochar alone (35% increase) and 2.3-fold higher than macrophyte alone (27% increase). Bulk density in M+B decreased by 22% (from 1.51 g/cm^3 to 1.18 g/cm^3), while total porosity increased by 38% (from 37% to 51%).

Nutrient removal efficiency was highest in M+B: soil TN reduced by 48% (from 5.2 g/kg to 2.7 g/kg), and water TN reduced by 65% (from 8.5 mg/L to 3.0 mg/L). This was 1.6-fold higher than biochar alone (30% soil TN reduction) and 1.9-fold higher than macrophyte alone (25% soil TN reduction). TP removal showed similar trends: M+B reduced soil TP by 42% and water TP by 58%, outperforming single treatments.

Heavy metal bioavailability was significantly reduced by M+B: DTPA-extractable Cd and Pb decreased by 71% and 68%, respectively. In contrast, biochar alone reduced Cd and Pb bioavailability by 45% and 42%, while macrophyte alone had minimal effect (18% and 15% reduction).

Microbial functional recovery was most pronounced in M+B: Shannon-Wiener index increased by 38% (from 2.6 to 3.6), and AWCD increased by 52% (from 0.39 to 0.59). Functional gene abundances also recovered: *nirS* and *pmoA* increased by 2.1-fold and 1.8-fold, respectively, compared to the control.

Table 2. Effects of remediation treatments on wetland soil and water properties (mean ± standard deviation)

Treatment	Ksat (cm/d)	Bulk Density (g/cm³)	Soil TN (g/kg)	Water TN (mg/L)	DTPA-Cd (mg/kg)	DTPA-Pb (mg/kg)	Shannon-Wiener Index
CK	8.0 ± 0.7	1.51 ± 0.08	5.2 ± 0.4	8.5 ± 0.6	1.2 ± 0.1	145 ± 12	2.6 ± 0.2
B	10.8 ± 0.9**	1.32 ± 0.07**	3.6 ± 0.3**	5.8 ± 0.5**	0.66 ± 0.06**	84 ± 9**	3.0 ± 0.2**
M	10.16 ± 0.9**	1.38 ± 0.07**	3.9 ± 0.3**	6.2 ± 0.5**	0.98 ± 0.08**	123 ± 11**	2.8 ± 0.2**
M+B	12.96 ± 1.1**	1.18 ± 0.06**	2.7 ± 0.3**	3.0 ± 0.4**	0.35 ± 0.04**	46 ± 6**	3.6 ± 0.2**
*(Note: *$p < 0.01$ compared to CK; n = 3 replicates)*							

4.4 Ecosystem Service Assessment

The M+B treatment achieved the highest total ecosystem service index (4.7), outperforming B (3.2), M (2.8), and CK (1.0) (Table 3). Hydrological regulation service was highest in M+B (index: 4.8) due to increased Ksat (12.96 cm/d) and WHC (32% higher than CK). Nutrient removal service (index: 4.6) was driven by high TN/TP reduction efficiency, while carbon sequestration service (index: 4.7) was attributed to SOC accumulation (0.8 tons C/ha over 120 days)—1.9-fold higher than CK.

In contrast, biochar alone had a lower hydrological regulation index (3.5) due to limited porosity improvement, and macrophyte alone had a low carbon sequestration index (2.5) because of slow SOC accumulation. The M+B treatment's synergy between biochar (soil structure improvement) and macrophytes (nutrient uptake, microbial stimulation) maximized ecosystem service provision.

Table 3. Ecosystem service assessment of remediation treatments (index: 1–5, 5 = highest service)

Treatment	Hydrological Regulation	Nutrient Removal	Carbon Sequestration	Total Service Index
M+B	4.8	4.6	4.7	4.7
B	3.5	3.2	2.9	3.2
M	2.7	3.0	2.5	2.8
CK	1.0	1.0	1.0	1.0

5. Discussion

5.1 Drivers of Suburban Wetland Soil Health Degradation

This global study identifies three interrelated drivers of urbanization-induced wetland soil degradation:

Hydrological alteration: High impervious surface cover (ISC > 30%) reduces groundwater recharge and increases stormwater runoff, leading to soil compaction (bulk density +38%) and reduced Ksat (-45%) . Compaction limits macroporosity, creating anaerobic conditions that enhance methane emission and metal mobilization—explaining the positive correlation between bulk density and Cd bioavailability ($r = 0.68$, $p < 0.01$).

Nutrient enrichment: Wastewater discharge and stormwater runoff increase TN by 3.8-fold in high-urbanization wetlands. Excess nutrients favor algal blooms, which decompose anaerobically and reduce oxygen levels—suppressing oligotrophic microbes and lowering microbial functional diversity (-32%) .

Heavy metal contamination: Traffic-related Pb (brake wear) and industrial Cd (metal plating) accumulate in high-urbanization wetlands, with concentrations exceeding SQGs in 62% of sites. HM stress disrupts microbial enzyme activity, reducing denitrification gene (*nirS*) abundance by 52% and impairing nitrogen removal .

Regional differences in degradation drivers are notable: Cambridge (UK) wetlands show severe nutrient enrichment (TN = 6.2 g/kg) due to outdated wastewater treatment, while Barcelona (Spain) wetlands have high Pb contamination (280 mg/kg) from dense traffic. This highlights the need for region-specific remediation strategies.

5.2 Synergistic Mechanisms of Macrophyte-Biochar Remediation

The M+B treatment's superior performance stems from three synergistic effects:

Soil structure improvement: Reed biochar (pyrolyzed at 600°C) has a high specific surface area (210 m^2/g) and porous structure, reducing bulk density by 22% and increasing Ksat by 62% . Biochar's macro-pores (50–100 μm) enhance water infiltration, while its surface functional groups (carboxyl, hydroxyl) adsorb HMs—reducing Cd/Pb bioavailability by 71%/68%.

Nutrient cycling restoration: *Vallisneria natans* absorbs nutrients via roots (removing 48% soil TN) and secretes oxygen into the rhizosphere, creating aerobic micro-zones that stimulate nitrifying/denitrifying microbes . This increases *nirS* gene abundance by 2.1-fold, enhancing nitrogen removal efficiency.

Microbial habitat optimization: Biochar provides a refuge for microbes against HM stress, while macrophyte root exudates (organic acids, amino acids) serve as carbon sources—increasing microbial functional diversity by 38%. The synergy between biochar (physical support) and macrophytes (resource provision) addresses the limitations of single treatments.

5.3 Implications for Suburban Wetland Management

Based on global results, we propose three targeted management strategies:

High-urbanization wetlands (Nanjing, Barcelona): Prioritize M+B remediation. Supplement with stormwater retention systems (e.g., permeable pavements) to reduce ISC and compaction. For Pb-contaminated sites (Barcelona), add iron-modified biochar to enhance Pb immobilization.

Moderate-urbanization wetlands (Sendai, Atlanta): Implement preventive measures—e.g., buffer zones between wetlands and urban areas to reduce nutrient/HM input. For mild degradation, single biochar amendment is sufficient to improve Ksat and SOC.

Nutrient-enriched wetlands (Cambridge): Combine M+B with enhanced wastewater treatment (e.g., constructed wetlands) to reduce TN input. Promote native macrophytes (e.g., *Iris pseudacorus* in UK) to enhance ecological compatibility.

5.4 Limitations and Future Research

This study has three limitations: (1) The remediation experiment was conducted in controlled greenhouse conditions—field validation is needed to assess long-term (3+ years) efficiency under natural hydrological fluctuations; (2) Sampling focused on temperate wetlands—tropical wetlands (e.g., Southeast Asia) may have different degradation patterns due to high temperature and rainfall; (3) The study did not evaluate the impact of M+B on aquatic biodiversity (e.g., invertebrates, fish).

Future research should: (1) Conduct multi-year field trials in different climatic zones; (2) Explore the effect of M+B on higher trophic levels; (3) Optimize biochar pyrolysis temperature and macrophyte density for region-specific conditions.

6. Conclusions

This global comparative study (UK, China, Spain, Japan, USA) systematically characterized suburban wetland soil health degradation under urbanization and validated a novel ecological remediation technology. Key findings include:

Degradation patterns: High-urbanization wetlands show 45% lower Ksat, 3.8-fold higher TN, 2.5-fold higher HM concentrations, and 32% lower microbial functional diversity than low-urbanization wetlands—driven by hydrological alteration, nutrient enrichment, and HM contamination.

Remediation efficiency: Macrophyte-biochar (M+B) co-application outperforms single treatments, increasing Ksat by 62%, reducing TN by 48% and HM bioavailability by 68–71%, and enhancing microbial diversity by 38%.

Ecosystem services: M+B maximizes hydrological regulation, nutrient removal, and carbon sequestration, achieving a total ecosystem service index of 4.7—2.1-fold higher than traditional chemical remediation.

The M+B technology provides a scalable, eco-friendly solution for suburban wetland soil health restoration. Its ability to address hydrological, nutrient, and heavy metal degradation simultaneously, while enhancing ecosystem services, makes it suitable for global application. Region-specific adjustments (e.g., iron-modified biochar for Pb-contaminated sites, native macrophytes for ecological compatibility) further ensure its adaptability across diverse urbanization contexts. This study advances our understanding of urban wetland degradation mechanisms and offers actionable tools to reconcile urban expansion with wetland ecosystem sustainability.

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