How cities lock in biodiversity persistence, recovery, and decline

Author
Allen Glen Cumaya Gil
Yale School of the Environment, Yale University
New Haven, Connecticut, United States
Email: allen.gil@aya.yale.edu
ORCID: 0000-0003-2347-296X

Preprint Statement

This manuscript is a non-peer-reviewed preprint submitted to EcoEvoRxiv and is currently under consideration for peer review at *npj Urban Sustainability*. It has not yet been certified by peer review. The author welcomes constructive feedback, comments, and suggestions that may help improve the manuscript.

Correspondence

Correspondence concerning this manuscript should be addressed to Allen Gil at allen.gil@aya.yale.edu.

Suggested Citation

Gil, A.G.C. ([year]). *How cities lock in biodiversity persistence, recovery, and decline*. EcoEvoRxiv preprint. [DOI]

Perspective

How cities lock in biodiversity persistence, recovery, and decline

Allen Glen Gil[1]

[1]Yale School of the Environment, Yale University, New Haven, Connecticut, 06511, United States

Abstract

Cities are expanding biodiversity plans, restoration projects, green infrastructure, corridors, and nature-based solutions. This Perspective defines biodiversity lock-ins as self-reinforcing urban pathways that make it difficult to reverse biodiversity persistence, recovery, or decline. It contributes a durability lens that links six urban mechanisms with biodiversity-specific features, including spatial dependency, ecological delay, recovery thresholds, and teleconnected impacts. The framework identifies intervention windows, keystone mechanisms, and strategies for redirecting urban systems toward durable biodiversity persistence and recovery.

Keywords: Biodiversity lock-ins; urban biodiversity; path dependence; social–ecological systems; nature-positive cities

Introduction

Cities are expanding biodiversity plans, restoration projects, corridors, green infrastructure, and nature-based solutions (Garrard et al., 2018; Nilon et al., 2017; Xie & Bulkeley, 2020). Their long-term value depends on whether they change the systems that shape biodiversity outcomes over time. Roads, drainage networks, zoning codes, land markets, supply chains, cultural expectations, ecological feedbacks, and monitoring indicators can reinforce biodiversity decline even as cities become greener (Buzási & Csizovszky, 2023; Lenzen et al., 2012; Moran & Kanemoto, 2017; Seto et al., 2016; Xie & Bulkeley, 2020).

Urbanization can drive habitat loss, fragmentation, biotic homogenization, and ecological filtering (Aronson et al., 2014, 2016; Fahrig, 2003; Haddad et al., 2015; Kowarik, 2011; McPhearson et al., 2016). It can also create habitats that support some species (Aronson et al., 2014, 2016; Beninde et al., 2015; Fahrig, 2003; Haddad et al., 2015; Kowarik, 2011; Lepczyk et al., 2017). Urban biodiversity also supports conservation, climate adaptation, ecosystem services, public health, environmental justice, and resilience (Haase et al., 2014; Kabisch et al., 2017; McPhearson et al., 2016; Wolch et al., 2014).

Urban biodiversity debates have paid less explicit attention to the durability of biodiversity outcomes over time (Buzási & Csizovszky, 2023; Nilon et al., 2017; Xie & Bulkeley, 2020).

Biodiversity outcomes become durable when infrastructure, institutions, finance, socio-cultural systems, ecological feedbacks, and knowledge systems reinforce one another over time (Buzási & Csizovszky, 2023; McPhearson et al., 2016; Seto et al., 2016). Greening can increase vegetation cover while leaving habitat fragmentation unresolved (Fahrig, 2003; Haddad et al., 2015). Local restoration can coexist with supply chains that drive distant biodiversity loss (Lenzen et al., 2012; Moran & Kanemoto, 2017). Cities can also make restored habitats, corridors, native species, stewardship systems, and ecological monitoring difficult to dismantle (Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017). The key question is whether urban systems make biodiversity gains durable, fragile, reversible, or increasingly difficult to redirect.

This Perspective defines biodiversity lock-ins as self-reinforcing urban pathways that make biodiversity persistence, recovery, or decline difficult to reverse. The concept draws from carbon lock-in, path dependence, socio-technical transitions, biodiversity governance, ecological resilience, extinction debt, and telecoupling (Geels, 2011; Halley et al., 2016; Jia et al., 2023; Liu et al., 2013; Pierson, 2000; Seto et al., 2016). These literatures explain important parts of the problem: self-reinforcing systems, institutional persistence, transition pathways, governance gaps, ecological thresholds, delayed losses, and distant impacts. Biodiversity lock-in brings these insights together around one question: which urban systems make biodiversity outcomes durable, reversible, or increasingly difficult to redirect? Its added value lies in integration: it links urban mechanisms to biodiversity-specific features that shape persistence, recovery, and decline across local and distant landscapes. It also identifies keystone mechanisms that provide leverage to redirect negative pathways and secure positive ones.

How biodiversity lock-ins work

Biodiversity lock-ins are shaped by five biodiversity-specific features. These features make biodiversity lock-ins ecologically distinctive because living populations, habitat relationships, and recovery thresholds respond unevenly across space and time. First, biodiversity is spatially relational. Species depend on place-specific relationships among habitat configuration, movement pathways, hydrology, nesting sites, host plants, edge conditions, disturbance regimes, and ecological interactions (Aronson et al., 2016; Beninde et al., 2015; Fahrig, 2003; Haddad et al., 2015). Habitat lost in one place cannot always be replaced elsewhere because ecological value depends on local context, habitat quality, species interactions, and landscape configuration (Beninde et al., 2015; Fahrig, 2003; Haddad et al., 2015). Urban systems can therefore lock in decline when roads, buildings, drainage networks, property boundaries, and land-use patterns sever relationships that are difficult to reconnect (Fahrig, 2003; Haddad et al., 2015; Seto et al., 2016).

Second, biodiversity is ecologically dependent. Outcomes depend on soils, hydrology, dispersal, species interactions, population viability, and recolonization (Beninde et al., 2015; Folke, 2006; Scheffer et al., 2001). Invasive species, heat stress, pollution, altered hydrology, degraded soils, and trophic simplification can push ecosystems into degraded states (Folke, 2006; Kowarik, 2011; Scheffer et al., 2001).

Third, biodiversity change often unfolds slowly. Habitat loss and fragmentation can commit species and communities to future decline before losses become visible (Halley et al., 2016). Ecological time lags can delay biodiversity responses after habitat change (Chen et al., 2023). Early gains in vegetation cover or species counts can therefore overstate the durability of recovery when delayed losses, ecological traps, or extinction debts remain unresolved (Battin, 2004; Chen et al., 2023; Halley et al., 2016; Schlaepfer et al., 2002).

Fourth, biodiversity pathways can cross recovery thresholds. Local extirpations, extinctions, genetic erosion, degraded soils, hydrological disruption, invasive dominance, and lost ecological interactions can reduce restoration potential (Haddad et al., 2015; Halley et al., 2016). Some losses make recovery slower, costlier, uncertain, or impossible (Haddad et al., 2015; Halley et al., 2016; Scheffer et al., 2001).

Fifth, urban biodiversity outcomes are teleconnected. Cities depend on food, timber, construction materials, energy, finance, logistics, and waste systems that connect urban life to land-use change elsewhere (Jia et al., 2023; Lenzen et al., 2012; Liu et al., 2013; McManamay et al., 2022; Moran & Kanemoto, 2017). A biodiversity lock-in lens, therefore, asks whether cities are improving local habitats and reducing wider systems that reproduce biodiversity loss elsewhere.

Negative and positive biodiversity lock-ins

Negative biodiversity lock-ins are self-reinforcing arrangements that reproduce habitat loss, fragmentation, ecological simplification, and displaced harm. They can emerge through fragmented urban form, biodiversity-blind infrastructure, weak ecological mandates, degraded ecosystems, extractive supply chains, and narrow accounting systems (Fahrig, 2003; Haddad et al., 2015; Halley et al., 2016; Jia et al., 2023; Shrestha et al., 2012; Xie & Bulkeley, 2020). These arrangements isolate habitats, weaken ecological processes, create delayed losses, reduce future recovery potential, and shift harm to distant landscapes (Fahrig, 2003; Haddad et al., 2015; Halley et al., 2016; Lenzen et al., 2012; Moran & Kanemoto, 2017).

Positive biodiversity lock-ins create durable support for persistence, recovery, adaptation, and stewardship. A corridor can become a positive biodiversity lock-in when zoning protects it, finance maintains it, residents value it, monitoring tracks ecological performance, and nearby habitats support its function (Beninde et al., 2015; Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017). The goal is to make persistence and recovery the most durable biodiversity pathways.

Six interacting mechanisms of biodiversity lock-in

Biodiversity lock-ins operate through six interacting mechanisms. Figure 1 illustrates how these mechanisms interact within urban systems, and Table 1 translates them into diagnostic questions for identifying biodiversity lock-ins. Infrastructure shapes habitat configuration, ecological flows, hydrology, edge effects, and restoration options (Fahrig, 2003; Haddad et al., 2015; Seto et al., 2016; Shrestha et al., 2012). Roads, drainage systems, ports, utilities, buildings, hardened waterways, parcels, and transport corridors can fragment habitat or support recovery when redesigned to reconnect habitats, restore hydrology, and reduce ecological barriers (Fahrig, 2003; Haddad et al., 2015; Seto et al., 2016; Shrestha et al., 2012). Infrastructure also links cities to distant biodiversity pressures through logistics, energy systems, construction materials, and waste systems (Lenzen et al., 2012; Moran & Kanemoto, 2017).

Institutions define rules and responsibilities. Zoning, permitting, environmental review, ownership systems, agency mandates, planning routines, and maintenance duties determine which ecological priorities become binding (Buzási & Csizovszky, 2023; Nilon et al., 2017; Xie & Bulkeley, 2020). Institutions make decline durable when they treat habitat as leftover land, divide authority, or omit long-term maintenance (Buzási & Csizovszky, 2023; Nilon et al., 2017; Xie & Bulkeley, 2020). They make recovery durable when biodiversity requirements enter zoning, development review, corridor protections, and adaptive management (Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017; Xie & Bulkeley, 2020).

Finance shapes incentives and resources. Framing finance as a biodiversity lock-in mechanism extends carbon lock-in and telecoupling scholarship to urban biodiversity: capital flows, public budgets, procurement, insurance, real estate finance, maintenance funding, and investment rules influence which land uses become profitable and which restoration projects persist (Seto et al., 2016). Decline becomes durable when fiscal systems reward ecological externalization or fund restoration without maintenance (Lenzen et al., 2012; Liu et al., 2013; McManamay et al., 2022; Moran & Kanemoto, 2017; Seto et al., 2016). Recovery becomes durable when budgets, procurement standards, and stewardship funds support ecological networks (Garrard et al., 2018; Kirk et al., 2021).

Socio-cultural systems shape accepted forms of urban nature. Preferences for low-density living, manicured landscapes, ornamental plants, artificial lighting, and controlled nature can simplify ecosystems (Aronson et al., 2016; Kowarik, 2011; Seto et al., 2016). Ecological literacy, stewardship norms, attachment to native species, and acceptance of habitat complexity can support recovery (Lepczyk et al., 2017; Nilon et al., 2017). Diets, material preferences, waste practices, and consumption expectations also connect cities to distant biodiversity pressure (Jia et al., 2023; Lenzen et al., 2012; Matej et al., 2024; Moran & Kanemoto, 2017).

Ecological feedbacks reinforce decline, persistence, or recovery. Fragmentation, extirpations, invasive species, ecological traps, degraded soils, altered hydrology, heat stress, pollution, and trophic simplification can reduce population viability and restoration potential (Battin, 2004; Fahrig, 2003; Folke, 2006; Haddad et al., 2015; Halley et al., 2016; Schlaepfer et al., 2002). Recovery becomes durable when connected habitats, restored hydrology, native vegetation, recolonization, trophic interactions, and viable metapopulations reinforce ecological function (Beninde et al., 2015; Fahrig, 2003; Garrard et al., 2018; Haddad et al., 2015).

Knowledge systems shape what cities see, measure, interpret, and treat as success. Canopy cover, park area, or greenness scores may miss habitat quality, species persistence, genetic diversity, ecological traps, extinction debt, and distant impacts (Battin, 2004; Halley et al., 2016; Nilon et al., 2017; Schlaepfer et al., 2002; Xie & Bulkeley, 2020). Knowledge-system lock-ins also emerge when expert-led indicators exclude local ecological knowledge, traditional ecological knowledge, community observations, practitioner experience, and stewardship knowledge (Danielsen et al., 2009; Tengö et al., 2014). Recovery is better supported when scientific monitoring, participatory mapping, citizen science, local and traditional ecological knowledge, and co-produced indicators jointly track ecological performance and accountability (Danielsen et al., 2009; Norström et al., 2020; Tengö et al., 2014).

These mechanisms often reinforce one another. Infrastructure can fragment habitat. Institutions can permit or legitimize that fragmentation. Finance can fund projects that continue it. Socio-cultural expectations can make biodiversity loss seem normal or acceptable. Ecological feedbacks can make recovery harder over time. Knowledge systems can also fail to detect losses that appear slowly or occur beyond city boundaries (Buzási & Csizovszky, 2023; Seto et al., 2016). A biodiversity lock-in lens helps identify which mechanisms are keeping a biodiversity pathway in place, whether recovery is still possible, which mechanisms offer the strongest leverage, and where action can still change the trajectory. In this way, the framework brings self-reinforcement, delayed loss, reversibility, and teleconnected responsibility into one analytical view.

Keystone lock-in mechanisms are context-specific mechanisms that most strongly reinforce the current biodiversity pathway or provide the greatest leverage to redirect it. In negative lock-ins, they are priority targets for intervention to interrupt the decline. In positive lock-ins, they are key supports that should be protected and strengthened. Identifying these mechanisms moves the analysis from broad diagnosis to strategic action.

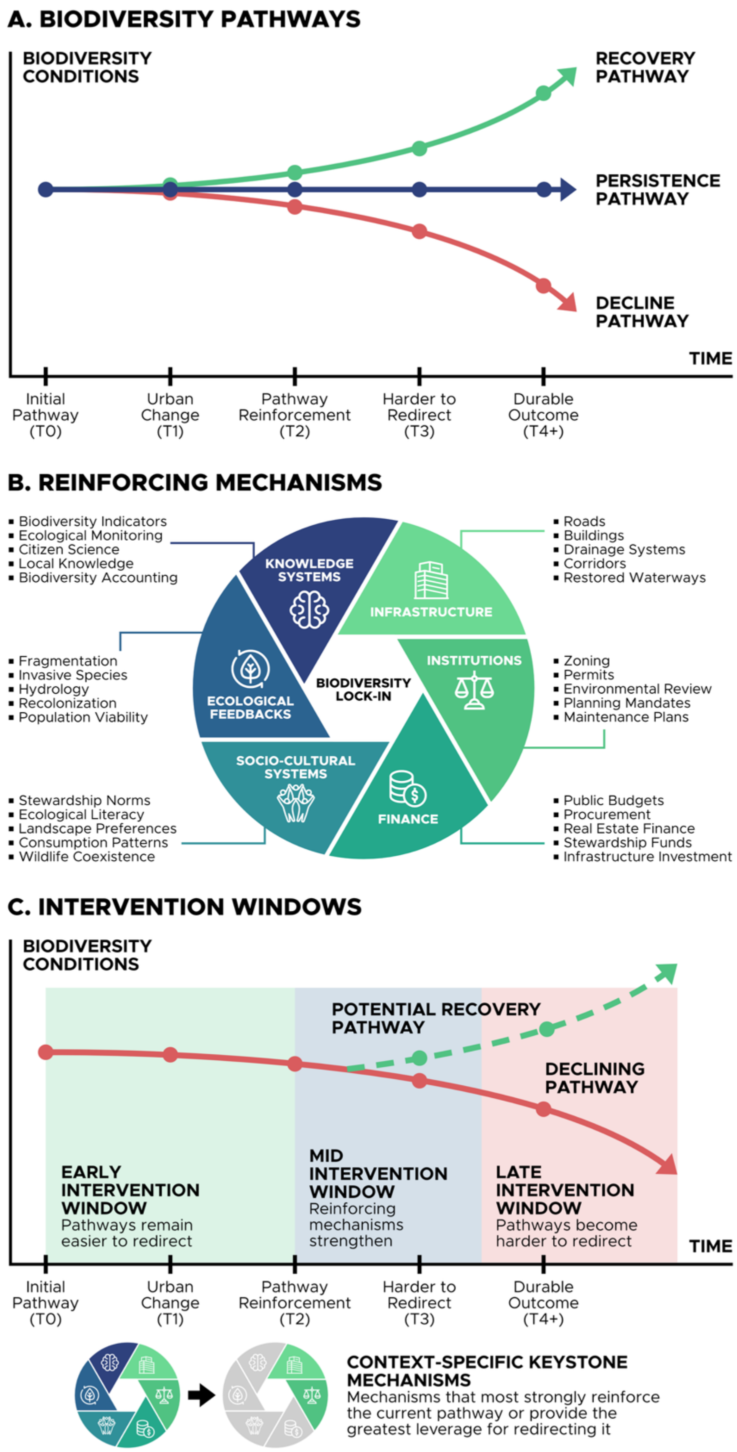

Figure 1. Biodiversity lock-ins in urban systems. Biodiversity lock-ins are self-reinforcing urban pathways that make it difficult to redirect biodiversity recovery, persistence, or decline. Panel A shows illustrative biodiversity pathways over time. Panel B shows six interacting mechanisms that reinforce or redirect these pathways. Panel C shows how intervention windows can be used to identify context-specific keystone mechanisms that provide leverage for redirecting biodiversity pathways.

Table 1 | Interacting mechanisms and diagnostic questions for biodiversity lock-ins

Mechanism	**What becomes durable**	**Negative lock-in**	**Positive lock-in**	**Diagnostic question**
Infrastructure	Habitat configuration and ecological flows	Built systems isolate habitat, disrupt hydrology, and limit reconnection	Corridors, crossings, restored waterways, and biodiversity-sensitive streetscapes support recovery	Do built systems connect or sever habitat relationships and ecological flows?
Institutions	Rules, authority, and responsibility	Weak zoning, fragmented mandates, and unclear duties normalize loss	Standards, corridor protections, maintenance duties, and coordination embed recovery	Do rules and responsibilities protect biodiversity over time?
Finance	Incentives and stewardship capacity	Investment, subsidies, and procurement externalize harm or underfund care	Stewardship funds, restoration finance, and biodiversity-sensitive procurement sustain care	Do financial flows support ecological performance and reduce displaced harm?
Socio-cultural systems	Norms, preferences, and stewardship	Manicured landscapes, ornamental species, and low tolerance for complex habitats simplify ecosystems	Ecological literacy, coexistence norms, and support for habitat complexity sustain recovery	Do public norms support complex, functional, and biodiverse habitats?
Ecological feedbacks	Population viability and recovery potential	Fragmentation, invasive species, degraded soils, and ecological traps reduce recovery potential	Native vegetation, restored hydrology, recolonization, and viable populations reinforce recovery	Are ecological feedbacks reinforcing decline, persistence, or recovery?

Knowledge systems	Visibility, interpretation, and accountability	Narrow indicators hide habitat quality, delayed loss, local knowledge, and distant impacts	Monitoring, citizen science, co-produced indicators, and supply-chain tracking support accountability	Do knowledge systems detect biodiversity quality, delayed effects, and teleconnected impacts?

Local and teleconnected biodiversity lock-ins

Biodiversity lock-ins operate within cities, across metropolitan regions, and through distant systems that support urban life. Local and metropolitan lock-ins shape habitat configuration, ecological dependency, governance, and restoration potential. Teleconnected lock-ins shape biodiversity outcomes elsewhere through food, energy, timber, minerals, construction materials, logistics, finance, and waste systems (Jia et al., 2023; Lenzen et al., 2012; Liu et al., 2013; McManamay et al., 2022; Moran & Kanemoto, 2017). Durable recovery therefore requires local habitat gains and reduced external ecological pressure.

The six mechanisms help make these dependencies visible. Infrastructure connects cities to distant landscapes through ports, logistics corridors, energy systems, and waste networks (Lenzen et al., 2012; Liu et al., 2013; McManamay et al., 2022; Moran & Kanemoto, 2017). Institutions shape distant impacts through procurement, certification, trade rules, and reporting standards (Jia et al., 2023; Liu et al., 2013; Matej et al., 2024). Finance links urban growth to commodity markets, infrastructure investment, and land-use change (Lenzen et al., 2012; McManamay et al., 2022; Moran & Kanemoto, 2017; Seto et al., 2016). Socio-cultural systems connect diets, housing expectations, material preferences, and consumption patterns to distant ecological pressure (Jia et al., 2023; Lenzen et al., 2012; Matej et al., 2024; Moran & Kanemoto, 2017). Knowledge systems determine whether these impacts appear in urban biodiversity accounting (Moran & Kanemoto, 2017; Nilon et al., 2017; Xie & Bulkeley, 2020).

In Vienna, diets with fewer animal products, avoiding food waste, following recommended caloric intake, and shifting from imports to domestic production could reduce the city's food-related biodiversity footprint by 21–43%, 5%, 9%, and 5–21%, respectively, depending on diet and demand (Matej et al., 2024). A biodiversity lock-in lens extends telecoupling research by focusing on durability: whether distant impacts are repeatedly reproduced through procurement, infrastructure materials, indicators, and supply chains.

Two illustrative city pathways

Two city pathways show how the framework can diagnose entrenched decline and embed recovery before urban form becomes difficult to change. Phoenix illustrates a negative biodiversity lock-in through metropolitan fragmentation. Fishermans Bend illustrates a prospective positive biodiversity lock-in through biodiversity-sensitive urban design. The cases also show how different mechanisms can become keystone depending on the pathway.

Phoenix Metropolitan Area: negative biodiversity lock-in through fragmentation

Rapid urbanization in Phoenix has been linked to population dynamics, water provisioning, transport technology, institutions, and topography (Shrestha et al., 2012). Interpreted through the biodiversity lock-in framework, these drivers form a reinforcing system that can make habitat fragmentation durable. Infrastructure appears to be a keystone mechanism in this case because roads, water systems, transport networks, and parcelization structure where development expands and where habitat reconnection remains possible (Seto et al., 2016; Shrestha et al., 2012). Built development separates habitats into smaller and more isolated patches, reshaping movement routes, disturbance gradients, hydrological conditions, and viable patch networks (Fahrig, 2003; Haddad et al., 2015; Shrestha et al., 2012).

Institutions, finance, socio-cultural expectations, ecological feedbacks, and knowledge systems reinforce this pathway. Land-use rules, jurisdictional fragmentation, infrastructure provision, development approvals, land markets, and real estate incentives can reproduce peripheral expansion (Seto et al., 2016; Shrestha et al., 2012). Preferences for low-density living, automobile access, private yards, and controlled landscapes can normalize development into desert ecosystems (Aronson et al., 2016; Seto et al., 2016). Fragmentation can reduce movement, increase edge effects, isolate populations, and limit recolonization (Fahrig, 2003; Haddad et al., 2015). Greenness, canopy, or open-space metrics may miss habitat quality, species persistence, ecological traps, or delayed decline (Battin, 2004; Halley et al., 2016; Nilon et al., 2017; Schlaepfer et al., 2002; Xie & Bulkeley, 2020). Targeting the infrastructure-development nexus could help redirect fragmentation before remaining reconnection options close.

Fishermans Bend, Melbourne: prospective positive lock-in through biodiversity-sensitive urban design

Fishermans Bend is one of Australia’s largest urban renewal projects and one of the first at this scale to include biodiversity targets through Biodiversity Sensitive Urban Design (Kirk et al., 2021). A biodiversity lock-in lens treats this case as a test of whether biodiversity goals can become durable through the systems that shape urban development. Streets, roofs, open spaces, water-sensitive systems, building interfaces, and public landscapes can be designed as habitat-supporting networks before parcels, buildings, drainage systems, and maintenance regimes become difficult to change (Garrard et al., 2018; Kirk et al., 2021).

Positive lock-in depends on enforceable rules, durable finance, public support, ecological function, and adaptive monitoring (Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017; Xie & Bulkeley, 2020). Biodiversity targets need to be connected to planning controls, development review, design guidelines, performance requirements, maintenance obligations, and adaptive management (Garrard et al., 2018; Kirk et al., 2021). Budgets, developer obligations, procurement rules, and stewardship arrangements need to support habitat features after construction (Garrard et al., 2018; Kirk et al., 2021). Public engagement and ecological literacy can normalize native vegetation, habitat complexity, standing water, dead wood, dense planting, and seasonal change (Garrard et al., 2018; Lepczyk et al., 2017; Nilon et al., 2017). Monitoring should combine scientific indicators, participatory monitoring, local ecological knowledge, stewardship feedback, and co-produced evaluation (Danielsen et al., 2009; Norström et al., 2020; Tengö et al., 2014). In this pathway, institutional rules, stewardship finance, and adaptive monitoring are likely keystone mechanisms because they determine whether biodiversity-sensitive design remains functional over time.

Operationalizing biodiversity lock-ins

Operationalizing biodiversity lock-ins means diagnosing whether an urban biodiversity pathway is being actively reproduced and becoming harder to reverse. The diagnostic steps synthesize insights from research on lock-in, path dependence, urban sustainability, ecological delay, and telecoupling (Buzási & Csizovszky, 2023; Halley et al., 2016; Liu et al., 2013; Pierson, 2000; Seto et al., 2016). The aim is to produce a diagnostic profile: what trajectory is being reinforced, which mechanisms sustain it, which mechanisms provide leverage, how reversible it remains, who holds responsibility, and where intervention remains possible. Biodiversity lock-in should be assessed using multiple lines of evidence and not reduced to a single score.

A pathway should be treated as a lock-in when three conditions are present: a consistent biodiversity trajectory, two or more reinforcing mechanisms, and narrowing options for reversal. The first diagnostic question is: what biodiversity trajectory is being reinforced? This may include persistence, recovery, simplification, decline, or displaced harm. Evidence may include habitat extent, connectivity, species occupancy, abundance, native species composition, functional diversity, habitat quality, restoration performance, ecological traps, extinction debt, and biodiversity footprint accounts.

The second question is: which mechanisms reproduce the trajectory? Planning documents, zoning codes, permits, agency mandates, infrastructure plans, budgets, procurement rules, maintenance contracts, investment patterns, stewardship practices, public preferences, ecological feedbacks, monitoring indicators, community observations, local ecological knowledge, participatory mapping, and co-produced indicators can show how biodiversity outcomes are built, authorized, funded, normalized, reinforced, or made visible. This step should also identify keystone mechanisms whose disruption, redesign, or reinforcement would have the greatest effect on the pathway.

The third question is: how difficult is reversal becoming? Reversibility has spatial, ecological, institutional, and financial dimensions. Spatial reversibility concerns habitat reconnection and restoration. Ecological reversibility concerns species, soils, hydrology, ecological interactions, and population processes. Institutional reversibility concerns rules, permits, responsibilities, and planning commitments. Financial reversibility concerns budgets, contracts, land values, and investment incentives.

The fourth question is: where can the pathway still be redirected? Intervention windows may appear before land is subdivided, corridors are blocked, permits are granted, infrastructure is built, maintenance duties are assigned, public expectations harden, invasive species dominate, or monitoring systems define success too narrowly. Because biodiversity data are uneven and many effects are delayed or displaced, lock-in diagnosis should report uncertainty from missing species records, weak baselines, unclear supply chains, contested responsibility, excluded local knowledge, and uncertain thresholds.

Future directions for biodiversity lock-in research

Future research should turn biodiversity lock-in from a conceptual lens into an empirical and practical research program. First, studies should identify when biodiversity outcomes are becoming locked in. This means distinguishing temporary greening from durable recovery, apparent stability from genuine persistence, and short-term decline from self-reinforcing degradation. Doing so requires longitudinal evidence on species trends, habitat quality, landscape configuration, restoration performance, ecological traps, governance commitments, and teleconnected impacts.

Second, research should examine how lock-in mechanisms interact across scales. Comparative studies can identify which combinations of infrastructure, institutions, finance, socio-cultural systems, ecological feedbacks, and knowledge systems most strongly reinforce biodiversity decline, persistence, or recovery. They can also identify keystone mechanisms that either sustain the current pathway or provide leverage for redirecting it. This work should connect local and distant impacts by tracing how municipal decisions, land markets, infrastructure networks, procurement systems, supply chains, and consumption patterns shape biodiversity outcomes across distance.

Third, research should identify when and where intervention is most effective. Planning research can examine when corridor protection, infrastructure redesign, restoration finance, zoning reform, procurement change, participatory monitoring, co-produced indicators, and biodiversity accounting are most likely to shift trajectories. It should also identify which mechanisms need strengthening so that positive biodiversity pathways can persist amid political, financial, ecological, and social change.

Sustaining biodiversity persistence and recovery

Urban biodiversity action gains long-term value when it changes the systems that reproduce decline, persistence, or recovery. Greening projects, restoration programs, corridors, and biodiversity targets become more durable when supported by infrastructure, institutions, finance, socio-cultural expectations, ecological feedbacks, and knowledge systems (Buzási &

Csizovszky, 2023; Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017; Seto et al., 2016). Sustaining persistence and recovery requires disrupting mechanisms that fragment habitats, weaken ecological function, and shift biodiversity harm elsewhere (Fahrig, 2003; Haddad et al., 2015; Lenzen et al., 2012; Moran & Kanemoto, 2017). It also requires design standards, stewardship finance, ecological monitoring, public support for habitat complexity, and institutions that sustain ecological repair (Garrard et al., 2018; Kirk et al., 2021; Nilon et al., 2017; Seto et al., 2016).

The central task is to redesign urban systems so that the persistence and recovery of biodiversity are easier to sustain. A biodiversity lock-in lens helps identify which pathways are being reinforced, which mechanisms are driving them, which mechanisms provide leverage, and where intervention can still change direction. Its core contribution is a durability lens for urban biodiversity: it evaluates whether urban systems make biodiversity persistence, recovery, or decline more likely to recur over time. This lens adds value by connecting local habitat configuration, ecological delay, recovery thresholds, governance commitments, finance, monitoring, and teleconnected harm into a single framework for diagnosing urban biodiversity futures.

Funding

The author received no specific funding for this work.

Data availability

No datasets were generated or analyzed for this Perspective.

Author contributions
A.G.G. conceptualized and wrote the manuscript.

Competing interests

The author declares no competing interests.

References

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Louwe Kooijmans, J., Kühn, I., MacGregor-Fors, I., McDonnell, M., Mörtberg, U., Pyšek, P., Siebert, S., Sushinsky, J., & Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. Proceedings of the Royal Society B: Biological Sciences, 281(1780), Article 20133330. https://doi.org/10.1098/rspb.2013.3330

Aronson, M. F. J., Nilon, C. H., Lepczyk, C. A., Parker, T. S., Warren, P. S., Cilliers, S. S., Goddard, M. A., Hahs, A. K., Herzog, C., Katti, M., La Sorte, F. A., Williams, N. S. G., & Zipperer, W. (2016). Hierarchical filters determine community assembly of urban species pools. Ecology, 97(11), 2952–2963. https://doi.org/10.1002/ecy.1535

Battin, J. (2004). When good animals love bad habitats: Ecological traps and the conservation of animal populations. Conservation Biology, 18(6), 1482–1491. https://doi.org/10.1111/j.1523-1739.2004.00417.x

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: A meta-analysis of factors determining intra-urban biodiversity variation. Ecology Letters, 18(6), 581–592. https://doi.org/10.1111/ele.12427

Buzási, A., & Csizovszky, A. (2023). Urban sustainability and resilience: What the literature tells us about "lock-ins"? Ambio, 52(3), 616–630. https://doi.org/10.1007/s13280-022-01817-w

Chen, X., Wang, Q., Cui, B., Chen, G., Xie, T., & Yang, W. (2023). Ecological time lags in biodiversity response to habitat changes. Journal of Environmental Management, 346, Article 118965. https://doi.org/10.1016/j.jenvman.2023.118965

Danielsen, F., Burgess, N. D., Balmford, A., Donald, P. F., Funder, M., Jones, J. P. G., Alviola, P., Balete, D. S., Blomley, T., Brashares, J., Child, B., Enghoff, M., Fjeldså, J., Holt, S., Hübertz, H., Jensen, A. E., Jensen, P. M., Massao, J., Mendoza, M. M., Ngaga, Y., Poulsen, M. K., Rueda, R., Sam, M., Skielboe, T., Stuart-Hill, G., Top-Jørgensen, E., & Yonten, D. (2009). Local participation in natural resource monitoring: A characterization of approaches. Conservation Biology, 23(1), 31–42. https://doi.org/10.1111/j.1523-1739.2008.01063.x

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. Annual Review of Ecology, Evolution, and Systematics, 34, 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Folke, C. (2006). Resilience: The emergence of a perspective for social-ecological systems analyses. Global Environmental Change, 16(3), 253–267. https://doi.org/10.1016/j.gloenvcha.2006.04.002

Garrard, G. E., Williams, N. S. G., Mata, L., Thomas, J., & Bekessy, S. A. (2018). Biodiversity sensitive urban design. Conservation Letters, 11(2), Article e12411. https://doi.org/10.1111/conl.12411

Geels, F. W. (2011). The multi-level perspective on sustainability transitions: Responses to seven criticisms. Environmental Innovation and Societal Transitions, 1(1), 24–40. https://doi.org/10.1016/j.eist.2011.02.002

Haase, D., Larondelle, N., Andersson, E., Artmann, M., Borgström, S., Breuste, J., Gómez-Baggethun, E., Gren, Å., Hamstead, Z., Hansen, R., Kabisch, N., Kremer, P., Langemeyer, J., Rall, E. L., McPhearson, T., Pauleit, S., Qureshi, S., Schwarz, N., Voigt, A., Wurster, D., & Elmqvist, T. (2014). A quantitative review of urban ecosystem service assessments: Concepts, models, and implementation. Ambio, 43(4), 413–433. https://doi.org/10.1007/s13280-014-0504-0

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., Melbourne, B. A., Nicholls, A. O., Orrock, J. L., Song, D.-X., & Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. Science Advances, 1(2), Article e1500052. https://doi.org/10.1126/sciadv.1500052

Halley, J. M., Monokrousos, N., Mazaris, A. D., Newmark, W. D., & Vokou, D. (2016). Dynamics of extinction debt across five taxonomic groups. Nature Communications, 7, Article 12283. https://doi.org/10.1038/ncomms12283

Jia, Q., Jiao, L., Hu, Y., Lian, X., Tian, Y., Liu, X., & Zhang, H. (2023). Telecoupling indirect ecological impacts of urban expansion in China from the perspective of the food trade. Land Degradation & Development, 34(16), 4964–4976. https://doi.org/10.1002/ldr.4822

Kabisch, N., van den Bosch, M., & Lafortezza, R. (2017). The health benefits of nature-based solutions to urbanization challenges for children and the elderly: A systematic review. Environmental Research, 159, 362–373. https://doi.org/10.1016/j.envres.2017.08.004

Kirk, H., Garrard, G. E., Croeser, T., Backstrom, A., Berthon, K., Furlong, C., Hurley, J., Thomas, F., Webb, A., & Bekessy, S. A. (2021). Building biodiversity into the urban fabric: A case study in applying Biodiversity Sensitive Urban Design (BSUD). Urban Forestry & Urban Greening, 62, Article 127176. https://doi.org/10.1016/j.ufug.2021.127176

Kowarik, I. (2011). Novel urban ecosystems, biodiversity, and conservation. Environmental Pollution, 159(8–9), 1974–1983. https://doi.org/10.1016/j.envpol.2011.02.022

Lenzen, M., Moran, D., Kanemoto, K., Foran, B., Lobefaro, L., & Geschke, A. (2012). International trade drives biodiversity threats in developing nations. Nature, 486(7401), 109–112. https://doi.org/10.1038/nature11145

Lepczyk, C. A., Aronson, M. F. J., Evans, K. L., Goddard, M. A., Lerman, S. B., & MacIvor, J. S. (2017). Biodiversity in the city: Fundamental questions for understanding the ecology of urban green spaces for biodiversity conservation. BioScience, 67(9), 799–807. https://doi.org/10.1093/biosci/bix079

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., Hertel, T. W., Izaurralde, R. C., Lambin, E. F., Li, S., Martinelli, L. A., McConnell, W. J., Moran, E. F., Naylor, R., Ouyang, Z., Polenske, K. R., Reenberg, A., de Miranda Rocha, G., Simmons, C. S., Verburg, P. H., Vitousek, P. M., Zhang, F., & Zhu, C. (2013). Framing sustainability in a telecoupled world. Ecology and Society, 18(2), Article 26. https://doi.org/10.5751/ES-05873-180226

Matej, S., Kaufmann, L., Semenchuk, P., Dullinger, S., Essl, F., Haberl, H., Kalt, G., Kastner, T., Lauk, C., Krausmann, F., & Erb, K.-H. (2024). Options for reducing a city's global biodiversity footprint: The case of food consumption in Vienna. Journal of Cleaner Production, 437, Article 140712. https://doi.org/10.1016/j.jclepro.2024.140712

McManamay, R. A., Brinkley, C., Vernon, C. R., Raj, S., & Rice, J. S. (2022). Urban land teleconnections in the United States: A graphical network approach. Computers, Environment and Urban Systems, 95, Article 101822. https://doi.org/10.1016/j.compenvurbsys.2022.101822

McPhearson, T., Pickett, S. T. A., Grimm, N. B., Niemelä, J., Alberti, M., Elmqvist, T., Weber, C., Haase, D., Breuste, J., & Qureshi, S. (2016). Advancing urban ecology toward a science of cities. BioScience, 66(3), 198–212. https://doi.org/10.1093/biosci/biw002

Moran, D., & Kanemoto, K. (2017). Identifying species threat hotspots from global supply chains. Nature Ecology & Evolution, 1, Article 0023. https://doi.org/10.1038/s41559-016-0023

Nilon, C. H., Aronson, M. F. J., Cilliers, S. S., Dobbs, C., Frazee, L. J., Goddard, M. A., O'Neill, K. M., Roberts, D., Stander, E. K., Werner, P., Winter, M., & Yocom, K. P. (2017). Planning for the future of urban biodiversity: A global review of city-scale initiatives. BioScience, 67(4), 332–342. https://doi.org/10.1093/biosci/bix012

Norström, A. V., Cvitanovic, C., Löf, M. F., West, S., Wyborn, C., Balvanera, P., Bednarek, A. T., Bennett, E. M., Biggs, R., de Bremond, A., Campbell, B. M., Canadell, J. G., Carpenter, S. R., Folke, C., Fulton, E. A., Gaffney, O., Gelcich, S., Jouffray, J.-B., Leach, M., Tissier, M. L., Martín-López, B., Louder, E., Loutre, M.-F., Meadow, A. M., Nagendra, H., Payne, D., Peterson, G. D., Reyers, B., Scholes, R., Speranza, C. I., Spierenburg, M., Stafford-Smith, M, Tengö, M, van der Hel, S., van Putten, I., & Österblom, H. (2020). Principles for knowledge co-production in sustainability research. Nature Sustainability, 3, 182–190. https://doi.org/10.1038/s41893-019-0448-2

Pierson, P. (2000). Increasing returns, path dependence, and the study of politics. American Political Science Review, 94(2), 251–267. https://doi.org/10.2307/2586011

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., & Walker, B. (2001). Catastrophic shifts in ecosystems. Nature, 413(6856), 591–596. https://doi.org/10.1038/35098000

Schlaepfer, M. A., Runge, M. C., & Sherman, P. W. (2002). Ecological and evolutionary traps. Trends in Ecology & Evolution, 17(10), 474–480. https://doi.org/10.1016/S0169-5347(02)02580-6

Seto, K. C., Davis, S. J., Mitchell, R. B., Stokes, E. C., Unruh, G., & Ürge-Vorsatz, D. (2016). Carbon lock-in: Types, causes, and policy implications. Annual Review of Environment and Resources, 41, 425–452. https://doi.org/10.1146/annurev-environ-110615-085934

Shrestha, M. K., York, A. M., Boone, C. G., & Zhang, S. (2012). Land fragmentation due to rapid urbanization in the Phoenix metropolitan area: Analyzing the spatiotemporal patterns and drivers. Applied Geography, 32(2), 522–531. https://doi.org/10.1016/j.apgeog.2011.04.004

Tengö, M., Brondizio, E. S., Elmqvist, T., Malmer, P., & Spierenburg, M. (2014). Connecting diverse knowledge systems for enhanced ecosystem governance: The multiple evidence base approach. Ambio, 43(5), 579–591. https://doi.org/10.1007/s13280-014-0501-3

Wolch, J. R., Byrne, J., & Newell, J. P. (2014). Urban green space, public health, and environmental justice: The challenge of making cities "just green enough." Landscape and Urban Planning, 125, 234–244. https://doi.org/10.1016/j.landurbplan.2014.01.017

Xie, L., & Bulkeley, H. (2020). Nature-based solutions for urban biodiversity governance. Environmental Science & Policy, 110, 77–87. https://doi.org/10.1016/j.envsci.2020.04.002