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Review

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OXFORD

Drivers of insect decline: direct, indirect, and combined pressures

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Reductions in insect biodiversity are widely documented and result from numerous pressures that operate from local to global scales. Identifying and synthesizing information on these drivers has become a central focus of research over the past decade, yet existing reviews often emphasize a limited set of well-established threats. Here, we provide a policy-oriented review of the drivers of insect decline, structured explicitly around the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) assessment framework and accompanied by statements of confidence to facilitate direct uptake into biodiversity assessments. We integrate evidence on both widely recognized drivers and a broader suite of pressures that remain comparatively understudied. The impacts of large-scale drivers of insect decline, such as land-use change and climate change, are well established and broadly align with those seen to threaten other taxa. Yet, many other threats, such as insecticides and light pollution (among others), pose disproportionate harm to insects but are not fully understood and remain important areas of research. It is also the case that most insect populations are experiencing multiple stressors simultaneously, and the combined effects of these pressures remain unclear. Despite this uncertainty, many threats to insect biodiversity are largely understood, and we know enough to form broad expectations for future responses and to develop action plans.

Keywords: insect decline, biodiversity loss, drivers of decline, Anthropocene

Introduction

Insects are the most species-rich group of organisms on Earth, and reductions in global biodiversity have entailed a substantial loss of insect life. While our understanding of the state of the world's insects is incomplete, it is clear that many insects are at risk of extinction, and many will likely be extinct before they are ever known (Wagner 2020). While this staggering loss of biodiversity is intrinsically tragic, the decline of insects also poses many challenges for human societies. Insects are integral components of nearly all freshwater and terrestrial ecosystems and provide key ecosystem services, including pollination, seed dispersal, decomposition, freshwater quality regulation, and pest control, among many others (Grames et al. 2026). Additionally, insects are a direct component of many human societies across the globe that depend on them as food sources, for recreation, and for livelihoods through entomotourism and beekeeping (Ware et al. 2026). A reduction in insect biodiversity is a significant loss for humanity and the natural world.

Identifying the anthropogenic forces responsible for these declines is therefore an urgent priority. This topic, relative to some of the others in this issue, has received considerable attention over the past decade (Sánchez-Bayo and Wyckhuys 2019, Wagner 2020, Wagner et al. 2021, Wilson and Fox 2021, Boyle et al. 2025). While a portion of this work covers topics addressed in such publications, especially the widely recognized direct drivers of decline, we also consider lesser-discussed direct stressors and indirect factors. Because of this structure, some topics receive disproportionately large treatment relative to their presumed contribution to global insect decline and the size of the literature supporting them. Section lengths should not be interpreted as statements of significance, and the relative importance of drivers of insect decline has been addressed in other publications (Sánchez-Bayo and Wyckhuys 2019, Halsch et al. 2025, Vidal et al. 2025). Each section is intended to provide a brief introduction to the literature on a particular threat, summarize key ideas, and offer resources for further reading.

The structure of this review follows that of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) biodiversity assessment report, adopting its confidence and hierarchy of evidence framework (IPBES 2019). Throughout, we use this framework to assess both the quantity of evidence and the level of scientific agreement for each key piece of information. These key statements can either be supported by a large amount of evidence (generally large-scale reviews or meta-analyses) with high agreement (well established), by a large amount of conflicting results (unresolved), with limited evidence but high agreement in the scientific community that merits further study (established but incomplete), or lacking strong evidence and agreement, often in nascent areas of study (inconclusive). Whenever possible, we prioritize citing meta-analyses and reviews over primary literature, as these provide greater insight into the state of a topic and are more useful for further investigation. We begin with direct drivers, the forces that can be readily tied to impacts on insects. We then move to indirect-to-direct drivers, which are often considered primary threats and are at the same time linked to other direct drivers (e.g. agriculture), before ending with indirect drivers, which are only connected to insect outcomes through mediating factors. The primary objective is to provide a high-level, expansive summary for policymakers that clarifies the strength of evidence and highlights major knowledge gaps (Fig. 1).

Key takeaways

- The drivers of insect decline are broadly known, but further research will improve prediction and conservation.
- Some threats, like insecticides and light pollution, pose a greater risk to insects than other taxa.
- Large-scale stressors like land-use and climate change affect insects across the world.
- Local-scale stressors, such as invasive species, when relevant, can have devastating consequences.
- The impacts of combinations of stressors is largely unknown yet is the reality for many species.

Fig. 1. Key takeaways about the drivers of insect decline.

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Direct Drivers

The direct drivers of insect biodiversity loss are the immediate forces that disrupt insects across levels of biological organization, and these have been well discussed over the past decade. Any comprehensive list will include land-use change, climate change, pesticides, and invasive species, among others. While some of these threats to insects directly align with the IPBES categories for drivers of change, others are more specific, and in these cases, we have classified them into the IPBES category they most closely align with (Fig. 2). For instance, many types of pollution are relevant to insects, so this review presents pollution across multiple subsections. Further, some widely discussed drivers of insect decline are not included in the IPBES framework and are presented here as separate sections (disturbance, pathogens, and the combination of drivers).

Land-Use Change

Land-Use Change Is One of the Primary Drivers of Insect Declines Worldwide, Aligning With Its Status as a Leading Cause of Biodiversity Declines Across Taxa (Well Established)

Anthropogenic land-use changes encompass both the conversion of natural and semi-natural ecosystems to human-dominated states, such as agriculture and urban infrastructure, and the degradation and fragmentation of habitat through intensification of existing land-use, increased management, chemical inputs, and landscape homogenization (Davison et al. 2021). Historically, land-use change has been considered the most significant driver of global biodiversity (Foley et al. 2005, Newbold et al. 2015, Beckmann et al. 2019, Caro et al. 2022). Importantly, historic habitat loss may result in long-term extinction debts that may impact populations and communities for decades (Sang et al. 2010). Insects, as a group, are hyperdiverse and are often highly specialized to specific habitats, with limited dispersal ability, making them especially sensitive to alterations in habitat extent, connectedness, and quality.

Agriculturally Driven Habitat Loss Remains One of the Greatest Threats to Insect Biodiversity (Well Established)

The past 300 years have been characterized by large-scale land conversion, with approximately 39% of Earth's ice-free terrestrial area currently used for agriculture, making it by far the most significant driver of biodiversity loss (Ellis et al. 2010). Ongoing large-scale conversion of tropical forests to agriculture is associated with severe declines in insect diversity (Boyle et al. 2025). Given that tropical forests host the greatest diversity of insect species, most of which remain undescribed, tropical deforestation represents one of the most immediate threats to insect diversity (Wagner 2020). Outside of the tropics, substantial evidence links modern farming practices that became widespread after World War II to insect declines driven by both agricultural intensification and land abandonment (Robinson and Sutherland 2002, Habel et al. 2019, Wagner 2020). Traditionally, low-intensity farming often maintained a diverse mosaic of habitats that supported diverse insect assemblages, and abandonment of these practices in favor of modern monoculture production has led to declines in herbivorous and pollinating insects (Tscharntke et al. 2002, Uchida and Ushimaru 2014, Raven and Wagner 2021, Prangel et al. 2023). Such intensified agricultural landscapes are characterized by reduced plant diversity, increased cropland extent, and decreased habitat connectivity and consistently show reduced insect richness, abundance, and evenness (Landis et al. 2000, Attwood et al. 2008, Seibold et al. 2019, Hemberger et al. 2021, Estrada-Carmona et al. 2022, Outhwaite et al. 2022). Another major footprint of agricultural habitat loss is the conversion of land for livestock production. Grazing is not typically associated with large-scale habitat destruction, yet its impacts can still be consequential (Filazzola et al. 2020, Pendrill et al. 2022). One important exception is the tropics, where beef production is now considered a leading cause of rainforest destruction (zu Ermgassen et al. 2020). Grazing can have transformative effects on plant communities, with context-dependent, but often negative, impacts on aquatic and terrestrial insects that vary with plant communities and grazing intensity (Strand and Merritt 1999, Kruess and Tscharntke 2002, Wang and Hou 2025). There is ongoing concern that implementing practices that support biodiversity in agricultural settings in some parts of the world might necessitate an offsetting increase in agricultural intensification elsewhere (Balmford et al. 2018).

Urbanization Contributes to Land Conversion and Habitat Loss (Well Established)

Urbanization negatively affects overall insect diversity and contributes to general decreases in insect abundance and functional diversity (Fenoglio et al. 2020, Vaz et al. 2023, Sanllorente et al. 2025). However, insect life-history traits condition responses to urbanization and land-use change, and species-level responses are inherently variable. For example, habitat specialists, poor dispersers, and species with narrow thermal or dietary niches tend to decline most severely, while mobile generalists often fare relatively better (Habel et al. 2016, Piano et al. 2017, Merckx and Van Dyck 2019, Nicvert et al. 2026). Urbanized environments introduce additional light and chemical pollution and heat-island effects, which can further disrupt insect life histories and population dynamics (Owens et al. 2020, Fenoglio et al. 2021). An important consideration in urbanization is the land that is being built over, as urban environments are more commonly established in already-modified landscapes than in natural ones (Simkin et al. 2022).

While the Harmful Impacts of Direct Habitat Loss Are Well Known, Habitat Fragmentation Is More Complex (Well Established)

Many landscapes supporting exceptional insect diversity are naturally fragmented (e.g. true islands, alpine meadows, patchy grasslands). Within these environments, insects usually persist as metapopulations that depend on movement, rescue effects, and recolonization for long-term persistence (Hanski 1998). In contrast to this natural landscape pattern, anthropogenic fragmentation refers to additional subdivision of habitat beyond natural configurations, creating smaller, more isolated, and more edge-dominated patches. Though effects of anthropogenic fragmentation on biodiversity vary, it has been broadly shown to reduce effective population sizes, increase local extinction risk, slow recolonization, and destabilize metapopulation dynamics, even when total habitat area remains unchanged (Fletcher et al. 2018). For example, empirical studies on butterflies show that isolation and patch configuration strongly influence species persistence, with fragment occupancy, abundance, and metapopulation stability often determined more by patch-level isolation and quality than by total habitat area alone (Thomas et al. 2001, Wenzel et al. 2006, Öckinger and Smith 2007, MacDonald et al. 2022). These ecological effects are also closely linked with genetic processes, such as inbreeding

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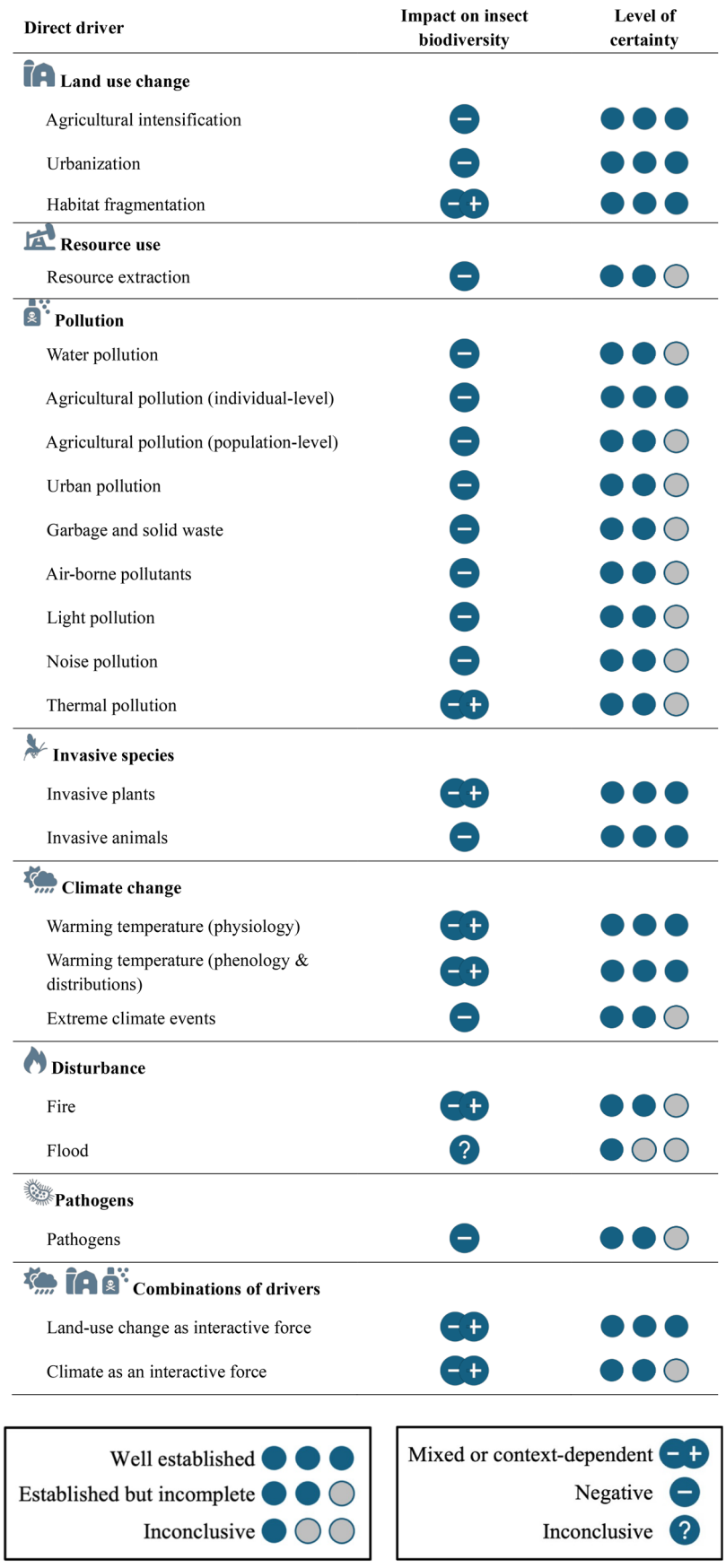


Fig. 2. Summary of the direct drivers of insect biodiversity change. Impact on insect biodiversity reflects the general direction of responses relative to an unmodified reference state.

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depression and loss of adaptive potential, and can drive populations into extinction vortices (Gilpin and Soulé 1986, Keller and Waller 2002). Species-level and trait-based analyses frequently reveal that fragmentation disproportionately impacts small-bodied, low-mobility, and host-specialist insects, resulting in reduced occupancy, lower abundance, and higher extinction probabilities within both individual small habitat fragments and entire fragmented landscapes (Öckinger et al. 2010, Fletcher et al. 2018, MacDonald et al. 2018). A comprehensive understanding of fragmentation effects, therefore, ultimately relies on autecological analyses that consider species-specific ecology and life history, a task made especially difficult by the incredible diversity and ecological heterogeneity of insects.

Resource Extraction

Resource Extraction Degrades Habitats, Especially Where Regulations Are Weak or Unenforced (Established but Incomplete)

The effects of extractive industries and their associated infrastructure are evident globally (Durán et al. 2013, Fernandes et al. 2024, Nakade and Dhadse 2024). In territories across North America, Europe, and Australia, extraction of oil, gas, coal, minerals, and water has caused altered and fragmented terrestrial and freshwater habitats, shifts in vegetation composition, and disruptions of soil microhabitats and edge effects, leading to a rise in pest populations and overall reductions in insect biodiversity (Finn and Knick 2011, Blinova and Dobrydina 2017, Curran et al. 2022, 2024, Fernandes et al. 2024). Extractive activities have also contributed to the pollution of waterways, which serve as habitat for many aquatic insects (Voss and Bernhardt 2017, Naslund et al. 2021), and the introduction of light pollution in otherwise dark areas (Longcore and Rich 2004). In the Global South, widespread mining and fossil fuel extraction have polluted and degraded terrestrial and aquatic ecosystems, causing increased insect pest and vector populations, reduced species and functional insect diversity, and homogenization of species assemblages across multiple biodiverse regions of South America (de Araújo et al. 2014, Enríquez Espinosa et al. 2020, Stoll et al. 2022, Espinosa et al. 2023), Africa (Ouma et al. 2022, Mulwanda et al. 2025), and Asia (Yapabandara et al. 2001, David 2003, Hasriyanty et al. 2022). In countries where environmental regulations are insufficient or unenforced, governments do not prioritize land reclamation, and extractive activities often extend into protected and biodiverse areas (Butt et al. 2013, Durán et al. 2013, Giam et al. 2018, Cabernard and Pfister 2022). However, countries with regulations mandating mitigation after extraction have seen positive responses from insect communities following restoration efforts (Buchori et al. 2018, Curran et al. 2022, 2024).

Water Pollution

Water Pollution Is a Threat to Aquatic Insects (Established but Incomplete)

Pollution of water refers to the input of nutrients, fertilizers, and pesticides (Liess et al. 2021), but also to less clearly visible forms of pollution such as microplastics (Eerkes-Medrano et al. 2015), trace metals (Zamora-Ledezma et al. 2021), pharmaceutical residuals such as antibiotics and hormones (Veseli et al. 2022), pollutants from roadside runoff (Soucek et al. 2025), and residues of industrial chemicals and other human-made products (Tang et al. 2025). Wastewater effluents affect the aquatic environment (water column, bottom substrate, and riparian vegetation) and, in turn, alter the behavior and physiological traits of aquatic insects. Runoff from insecticides poses a major threat, resulting in reductions in diversity and abundance (Beketov et al. 2013, Kellar et al. 2014, Barmentlo et al. 2025), individual mortality (Liess et al. 2021), decreased fitness (Chang et al. 2007, Tüzün and Stoks 2017), behavioral shifts such as reduced activity and foraging (Späth et al. 2022), and delayed larval development (Liu et al. 2018). Extractive activities have contributed to ionic imbalances in waterways and introduced pollutants, like heavy metals, that extirpate pollution-sensitive taxa, reduce macroinvertebrate biomass, and bioaccumulate in aquatic insects (Voss and Bernhardt 2017, Naslund et al. 2021). Bioaccumulation and bioamplification result in contaminant transfer from aquatic to terrestrial stages in holometabolic aquatic amphibiotic insects (Veseli et al. 2022, 2024). Many pollutants are more toxic to freshwater invertebrates under warming conditions; thus, the deleterious effects of water pollution may be stronger in landscapes experiencing significant climatic changes (Dinh et al. 2016, 2022, Macaulay et al. 2019).

Agriculture, Forestry, and Urban Pollution

Agricultural Pesticides Impact Non-target Species and Have Both Lethal and Sublethal Effects on Individuals (Well Established)

Pesticides, which include insecticides, fungicides, and herbicides, are widespread and affect non-target insects far from the original application (Nuyttens and Verboven 2015, Olaya-Arenas and Kaplan 2019, Halsch et al. 2020, Schmidt et al. 2022, Duchet et al. 2023, Lenard et al. 2025). For example, approximately 90% of neonicotinoids applied to seeds in the United States enter water or are taken up by non-crop plants (Krupke and Tooker 2020). Insecticides are specifically designed to target insects, and, as a result, have well-established lethal effects on key indicator taxa such as honey bees and many other non-target organisms. In cropped landscapes, insects are also often exposed to multiple compounds simultaneously, and the sublethal and combined effects of these compounds are less well understood (Tosi et al. 2022) and may be common (McArt et al. 2017, Graham et al. 2021, Mueller et al. 2024). A meta-analysis on the impacts of pesticides on non-target species revealed sublethal effects on animals (invertebrates and vertebrates) for growth, reproduction, and behavior across both terrestrial and aquatic systems (Wan et al. 2025). Another meta-analysis found that neonicotinoids negatively affected pollinator condition and reproductive success but had no significant effects on detritivores (Main et al. 2018). Neonicotinoids have received the most research and public attention; however, many other insecticides used to protect crops or cattle, such as pyrethroids, diamides, and organophosphates, are widely used and are known to impact a broad range of non-target insects (Gandara et al. 2024). Further still, fungicides, herbicides, and adjuvants also threaten insects, and further research is needed in this area (Ingram et al. 2015, Mullin et al. 2016, Shannon et al. 2023, Obregon et al. 2024).

The Impact of Pesticides on Individuals Scales up to the Population Level (Established but Incomplete)

Establishing population-level risks of pesticides includes the challenge of integrating not only lethal and sublethal toxicity but also actual exposure to these pesticides, along with other

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landscape-level stressors with which they often co-occur. For instance, insect diversity and abundance in Germany are lower near agricultural fields (Hallmann et al. 2017, Habel et al. 2019), but separating the effects of habitat loss from those of agricultural pesticides can be difficult. Bumble bee colonies across Europe have been shown to contain colony pollen with many types of pesticides, and colony production and colony weight decreased with increasing pesticide exposure (Nicholson et al. 2024). Observational studies suggest that there is a strong negative association between neonicotinoid seed treatments and wild bee extirpation rates in the UK (Woodcock et al. 2016), occupancy probability in the United States (Janousek et al. 2023, Guzman et al. 2024), and butterfly abundance and richness in the United States (Forister et al. 2016, Van Deynze et al. 2024). A meta-analysis of field experiments found that the abundance of insect predators is negatively affected by neonicotinoid-treated seed and by pyrethroid applications (Douglas and Tooker 2016), and some studies have linked area-wide patterns in neonicotinoid use to disruptions in the population cycling of biocontrol insects (Bahlai et al. 2015). The effects of other insecticides, fungicides, herbicides, and adjuvants are even less understood at the population level but are likely important given their ubiquity in some agricultural landscapes (Halsch et al. 2020). For example, there is evidence of synergistic interactions between insecticides and fungicides propagating through trophic interactions to reduce complexity in invertebrate communities within arable agricultural systems (Woodcock et al. 2025b). While establishing population-level consequences is more difficult than individual-level outcomes and remains an important area of research, it should be remembered that insecticides are specifically designed to harm insects via known physiological pathways and are widely found across landscapes (Fig. 3); as a result, they imply a more direct impact on insects relative to other threats.

Pesticides Are Found in Urban Spaces, and This Exposure Affects Insects (Established but Incomplete)

In contrast to monoculture cropping systems, the suite of compounds applied to urban landscapes is generally more varied (Wittmer et al. 2011), and beyond insecticides, fungicides, and herbicides, also includes rodenticides, miticides, ectoparasiticides, repellents, and fumigants (World Health Organization 2006). A global review of urban pesticide use found that these compounds are often applied indiscriminately, and that pesticide residues have been detected in urban soil, air, dust, and water (Meftaul et al. 2020). Pesticides in urban areas have been detected at concentrations that can cause lethal and sublethal effects within bees in North America (Siviter et al. 2023) and Europe (Cappellari et al. 2024, Šlachta et al. 2025), as well as in butterflies in North America and in the Amazon (Nowell et al. 2021, Rico et al. 2022, Byers et al. 2025, Dittemore et al. 2025). Urban pesticide runoff has also been implicated in reductions in alpha and beta diversity of macroinvertebrates in China (Hou et al. 2025) and in the displacement of aquatic macroinvertebrates in urban streams in the United States (Wurzel et al. 2020). Broadcast spraying of urban forests has been shown to reduce the abundance of beneficial parasitoids and predators (Quesada and Sadof 2020). Pesticide applications in urban gardens reduced butterfly and bumble bee richness and abundance (Muratet and Fontaine 2015, Tassin de Montaigu and Goulson 2024). Additionally, evidence is mounting that broadcast spraying of broad-spectrum insecticides for mosquito control can harm non-target insects (Oberhauser et al. 2006, Giordano et al. 2020, Wagner 2026). Research on pesticides in urban systems remains limited, and



Fig. 3. Selected anthropogenic threats that have disproportionate effects on insects compared to other taxa. (A) Insecticides are often applied across broad areas and are formulated to specifically disrupt insect physiology. (B) Commercial bees are transported worldwide, where they reshape pollinator competition and introduce diseases and pathogens into native insect communities. (C) Many insects are naturally attracted to light, and light pollution dramatically alters nighttime environments, increasing the risk of predation and excess energy expenditure. (D) Many insects are viewed as nuisances despite the ecosystem services they provide, and these perceptions hinder understanding and conservation. Photo credits: (A and B) Wikimedia Commons, (C) Brett Seymoure, (D) Wikimedia Commons.

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broader work using in situ, field-based methods to quantify exposure and risk will help separate the effects of pesticides from other stressors with which they often co-occur.

Garbage and Solid Waste

Garbage and Solid Waste Affect Insects, but Studies Are Limited (Established but Incomplete)

Solid waste releases leachates containing heavy metals and persistent organic pollutants, which impair insect development, reproduction, and survival (Huset et al. 2011, Liu et al. 2025). It can also modify insect communities by favoring some species through increased resource availability (e.g. food and habitat) while harming others through chemical contamination and physical habitat disruption (Qasim et al. 2020, Kheswa et al. 2025, McConnell et al. 2026). Within the study of solid waste, microplastics are the most researched and are now found in all terrestrial and aquatic environments (Zeb et al. 2024, Thrift et al. 2025). Microplastics are regularly ingested by insects (Gallitelli et al. 2024, Sucharitakul et al. 2024), and these occurrences are concentrated in areas of anthropogenic activity (Khdre et al. 2023, Rodrigues et al. 2024). Ingested plastics have been shown to interact with insects across different orders, with effects including blocking or damaging the digestive tract, interfering with cognition, reducing body weight, or being biofragmented and re-released (Viana et al. 2023, El Kholy et al. 2024, Pasquini et al. 2024, Ritchie et al. 2024). Microplastics also contain chemical additives or adsorb toxins that reduce insect fitness, but further study is needed to characterize ecologically relevant exposures (Shaha and Pandit 2020, Castilloux et al. 2022, Allison et al. 2026).

Air Pollution

Air Pollution Impairs Insect Physiology and Plant–Insect Interactions (Established but Incomplete)

Air pollution broadly refers to pollutants emitted into the atmosphere, ranging from compounds produced by burning fossil fuels to pesticides applied by aircraft. Given that pesticides were discussed in a previous pollution section, this section focuses on other airborne pollutants. Nitrogen is one of the world's leading air pollutants, and the nitrogen cycle has been reshaped over decades of change (Galloway et al. 2008). Nitrogen is a limiting nutrient in many ecosystems, and atmospheric sources of excess nitrogen can lead to nitrification, which has both direct and indirect effects on insect populations. Excess nitrogen has been shown to have severe negative impacts on insect abundance and diversity across the globe (Öckinger et al. 2006, Warren et al. 2021, Balfour et al. 2025, Fan et al. 2025), and such results are especially concerning given that some of the most species-rich plant communities across the planet occur on nutrient-poor soils that are at high risk of disruption (Wagner 2020). Other airborne pollutants, including tropospheric ozone, nitrogen oxides, sulfur dioxide, and particulate matter, contribute to insect biodiversity loss through multiple mechanisms, such as direct physiological toxicity, changes in host plant nutritional quality, and disruption of volatile organic compound-mediated interactions (Duque and Steffan-Dewenter 2024, Pinto-Zevallos et al. 2025). For example, experimental work found that pollinators and parasitoids suffer 21% to 35% declines in performance (e.g. abundance and search efficiency) in response to air pollution, while pest species showed little response (Ryalls et al. 2024), exacerbating imbalances in insect communities. Rising ozone levels (IPCC 2023, Chen et al. 2024, East et al. 2024) pose a significant threat to services such as pollination and natural pest regulation, impairing volatile organic compound detection even below regulatory thresholds (Pinto-Zevallos et al. 2025). Given the widespread nature and potential ecosystem impacts of atmospheric pollution, especially nitrification, this may be one of the most important and underappreciated direct drivers of insect decline.

Elevated CO_2 Changes Plant Quality and Alters Plant–Insect and Insect–Insect Interactions (Established but Incomplete)

Elevated atmospheric CO_2 can affect insect communities through increasing plant productivity but decreasing nutritional quality of plant tissues (Kaspari and Welti 2024). Altered plant nutrition under elevated CO_2 is expected to benefit sap-feeding herbivorous insects such as aphids and cicadas, but to have adverse effects on populations of chewing herbivorous insects such as grasshoppers and caterpillars (Robinson et al. 2012). Additionally, elevated CO_2 may alter interactions among insect guilds (Kremer et al. 2018) and between insects and plants (Filipiak et al. 2024). While the direct effects of elevated CO_2 on plants are well established, insect community responses have been observed in both experimental (Stiling et al. 1999) and observational settings in both Canada and the United States (Ziska et al. 2016, Welti et al. 2020), but much remains unresolved (Kaspari and Welti 2024).

Light Pollution

Light Pollution Alters the Natural Movements of Insects Via Positive or Negative Phototaxis as well as Their Natural Rhythms of Activity and Phenology (Established but Incomplete)

Light pollution is among the most rapidly growing anthropogenic threats, with sky brightness at night increasing at an estimated rate of 7% to 10% annually (Kyba et al. 2023). A single artificial light can attract hundreds or even thousands of insects per night, one-third of which are estimated to die before morning (Eisenbeis 2006), and there are likely over a billion outdoor light sources on Earth today. The effects of lighting on insects, via temporal, spatial, and spectral disorientation, have repeatedly been shown to significantly decrease fitness by altering growth and development, movement, foraging, reproduction, and predation risk (Owens et al. 2020, Boyes et al. 2021a). A field experiment in the UK found reductions in moth abundance of 47% and 33% in artificially lit hedgerows and fields compared to dark sites (Boyes et al. 2021b). The consequences of these losses for species persistence are just now being explored (Väisänen and Hublin 1983, González et al. 2026). Given that this driver is widely overlooked (Gaston 2019) and that light-based survey methods may be becoming less reliable for conservation monitoring (Battles et al. 2024), much remains to be done to quantify the severity of light pollution relative to other drivers (Fig. 3).

Noise Pollution

Noise Pollution Disrupts Insect Behavior and Physiology (Established but Incomplete)

Anthropogenic noise pollution has increased substantially over the past century in both terrestrial and aquatic environments (Barber et al. 2010, Hawkins et al. 2015). Increased noise is likely to negatively impact insects because noise pollution often differs from natural noise in frequency, duration, and

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patterning, and sometimes overlaps with the frequency ranges of insect songs and hearing (Morley et al. 2014). Noise pollution can disrupt acoustic communication, reproduction, foraging, danger avoidance, shift habitat use, change endocrine and metabolic systems, and alter biotic interactions (Römer 2013, Schmidt and Balakrishnan 2015, Raboin and Elias 2019, Classen-Rodríguez et al. 2021, Fenoglio et al. 2021). Thus far, observed effects include shifts in insect behavior, decreases in reproductive fitness, and reductions in abundance and diversity (Raboin and Elias 2019, Heinen-Kay et al. 2021, Anjana and Teji 2025). For example, experimental work shows that male grasshoppers raised near noisy roadsides produce different courtship songs than those raised in quiet environments (Lampe et al. 2014). While a growing body of research suggests significant effects of anthropogenic noise on insects, research remains limited and largely biased toward birds (Hawkins et al. 2015, Shannon et al. 2016, Sordello et al. 2020, Jerem and Mathews 2021, Gomes et al. 2022, Guenat and Dallimer 2023).

Thermal Pollution

Thermal Pollution Creates Novel Heated Environments and Has Mixed Effects on Insects (Established but Incomplete)

Compared with surrounding suburban and rural areas, terrestrial urban environments are typically warmer, with increased nighttime temperatures and reduced daily thermal fluctuations, which could disproportionately affect insects and other poikilotherms (Raupp et al. 2010, Speights et al. 2017, Gaston et al. 2023). Likewise, water systems in or near areas of high thermal pollution are typically warmer and experience heat fluxes from stormwater runoff and industrial cooling system thermal discharge (Hester and Doyle 2011, Somers et al. 2013, Quevedo et al. 2018, Wiederkehr et al. 2020). Insect responses to thermal pollution are diverse and include both increases and declines in insect abundance and diversity (Meineke et al. 2013, Miles et al. 2019, Wiederkehr et al. 2020, Fenoglio et al. 2021, Collins et al. 2024). These outcomes are the result of multiple mechanisms and include phenological changes altering biotic interactions with food sources and/or natural predators (Collins et al. 2024); creation of thermal refugia (Raupp et al. 2010), heat stress, changes in fecundity, and indirect effects from changes in plant physiology (Dale and Frank 2018, Miles et al. 2019). For example, in a field setting, wild bee species with higher critical thermal maximums responded more positively to urban heating than cooler-associated solitary and cavity-nesting bees (Hamblin et al. 2017). Overall, projections suggest that thermal pollution will significantly affect terrestrial and aquatic insects, but comprehensive reviews focused on thermal pollution are needed (Oertli and Parris 2019).

Invasive Species

Invasive Animals Tend to Have Substantial Adverse Effects on Native Insect Faunas (Well Established)

Globalization has greatly increased the frequency of accidental and intentional introductions of invasive species worldwide, often with dire consequences for insects (Sánchez-Bayo and Wyckhuys 2019, Skinner et al. 2026). This phenomenon has been particularly well studied in Hymenoptera (Montero-Castaño and Vilà 2012, Skinner et al. 2026). Invasive ants that were unintentionally introduced across the world severely affect native ants and can also reduce non-ant invertebrate abundance and species richness (Holway 1998, Lester and Tavite 2004, Savage et al. 2009, Vonshak et al. 2010, Stanley and Ward 2012, Cooling et al. 2015, Goodman and Warren II 2019, Zina et al. 2020, Devenish et al. 2021, Hasin et al. 2021). The negative impacts of invasive ants are disproportionately felt in island systems, where their introduction has led to cascades of biodiversity loss, from insects to birds (McGlynn 1999, Clavero et al. 2009, Wagner and Driesche 2010). In contrast, bees are often intentionally moved for pollination services and can compete with native bees for floral resources and nesting sites and can also serve as vectors of pathogens (Goulson 2003, Arismendi et al. 2021, Pasquali et al. 2025). When introduced bees compete with native bees, pollen and nectar availability can be reduced, leading to behavioral shifts in native bee communities (Page and Williams 2023, Pasquali et al. 2025). The roles of invasive bees as disease vectors are potentially devastating and contribute to bee decline (Colla et al. 2006, National Research Council 2007). Insects have also been moved for biocontrol purposes, as is the case with the harlequin ladybird, which has spread from Asia into Europe and North America, where it threatens native ladybird beetles (Roy and Brown 2015, Camacho-Cervantes et al. 2017, Perry et al. 2024). Non-insect invasive animals can also severely impact insect populations, as seen in aquatic systems where the introduction of crayfish has resulted in loss of larval and adult forms of aquatic taxa (McCarthy et al. 2006) and stocking of lakes with non-native sportfish can alter aquatic insect communities (Pope et al. 2009, Labat and Beisel 2025). The introduction of mammals into New Zealand offers some of the most tragic examples of extinction, famously of many birds, but such invasions have also had dramatic negative impacts on invertebrate communities (St Clair 2011, Houghton et al. 2019). In some instances, positive outcomes of animal invasion are documented, such as riffle beetles (*Stenelmis* spp.) that can benefit from invasive crayfish; however, these cases are largely exceptions (Freeland-Riggert et al. 2016).

Invasive Plant Species Often Negatively Affect Native Insects, but Their Impacts Are Context-Dependent and Can Include Benefits (Well Established)

Negative impacts of invasive plants on insects include reduced abundance and altered community composition (Topp et al. 2008, Wagner and Driesche 2010, Yoshioka et al. 2010, Schuh and Larsen 2015, Nguyen et al. 2016, Schirmel et al. 2016, Osbrink et al. 2018). Non-native plants have been incorporated into the diets of native herbivores (Graves and Shapiro 2003); however, these herbivores often perform worse on the introduced plant (Yoon and Read 2016). Here too, as with invasive animals, species on islands appear especially vulnerable. One stark example of this can be found in Hawaiian Lepidoptera: 28.8% of species are presumed extinct, largely due to invasive animals and plants (Austin and Rubinoff 2025). The impacts of invasive plants are not always negative, and invasive plants have increased food resources for carabid beetles (Hansen et al. 2009) and pollinators (Lopezaraiza-Mikel et al. 2007, Ramula and Sorvari 2017). For example, a third of California's butterflies are known to use introduced plants as larvae (Graves and Shapiro 2003). However, increases in the abundance of individual species do not necessarily improve ecosystem functioning. For example, higher pollinator numbers do not necessarily translate into better pollination (Lopezaraiza-Mikel et al. 2007).

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Climate Change

Insect Physiology Governs Responses to Changing Temperatures (Well Established)

As small ectotherms with large surface areas, insects are highly dependent on abiotic conditions, and there is already considerable evidence of a negative overall impact of climate change on insect biodiversity (Halsch et al. 2021, Menzel and Feldmeyer 2021, Wilson and Fox 2021, Harvey et al. 2023). To date, much of the research into climate change impacts has focused on climate warming. Temperature directly affects insect population dynamics and survival by impacting development, fecundity, reproduction, and dispersal (González-Tokman et al. 2020). Species living above their thermal optimum will likely experience declines with warming, while those below it will likely benefit up to a point. This can be seen in the decline of cold-adapted species and the increase of warm-adapted species in temperate Europe and North America (Termaat et al. 2019, Bowler et al. 2021, Shirey et al. 2024). Many tropical species are considered closer to their thermal limits (Deutsch et al. 2008, García-Robledo et al. 2016, Boyle et al. 2025, Holzmann et al. 2026) and therefore under greater threat from climate change, which is concerning given the high insect diversity in the tropics.

Warming Conditions Alter Insect Phenologies and Distributions (Well Established)

Phenological and distributional changes are well-documented responses to warming, and such shifts may allow species to offset some of the damaging impacts of climate change. Warming contributes to phenological shifts, including advances in springtime phenology (as well as delays), extended activity periods, changes to the number of generations per year, and changes in winter survival rates (Forrest 2016). Shifting phenology is a well-documented response to warming temperatures (Bartomeus et al. 2011, Forrest 2016), with multiple studies documenting shifts in phenology by many days per decade (Forister and Shapiro 2003, Cohen et al. 2018, Foster et al. 2026). Many species' ranges have shifted to higher latitudes (Chen et al. 2011, Kwon et al. 2014, Platts et al. 2019) or to higher elevations (Colwell et al. 2008, Kerr et al. 2015, McCain and Garfinkel 2021, Biella et al. 2024) in response to climate warming. Range shifts allow species to track their environmental niche and persist, but such shifts can be limited by habitat availability and dispersal distances (Grewe et al. 2013, Platts et al. 2019, Fourcade et al. 2021). These changes have been particularly evident in montane habitats (De Knijf et al. 2011, Halsch et al. 2021, McCain and Garfinkel 2021). Range shifts have also been well studied among pest species; notably, some bark beetle species have undergone warming-related range expansions in Europe and North America (Jaime et al. 2024). While predictable distributional shifts have been documented, many specialists do not occupy their entire fundamental niche (Stewart et al. 2015). In such species, biotic interactions may limit the capacity of phenological and distributional changes to offset the effects of warming.

Beyond Increases in Average Temperature, Other Aspects of Climate Change Have Negative Impacts on Insects (Established but Incomplete)

Many studies demonstrate the dramatic effects of changes in climate variables beyond mean temperature alone. Other changing aspects of climate include the increasing frequency and intensity of extreme heat events (Ma et al. 2021, Harvey et al. 2023), increased nighttime temperature (Speights et al. 2017, Halsch et al. 2021), and reduced snow depth and duration (Bale and Hayward 2010, Harris et al. 2019, Halsch et al. 2024). Increasingly, drought is being discussed as a major climatic threat, as insects are particularly vulnerable to changes in moisture due to their high surface area-to-volume ratio (Wagner 2020, Kazenel et al. 2024). Drought has been identified as a major cause of insect loss in both tropical and temperate systems (McDermott Long et al. 2017, Forister et al. 2018, Filazzola et al. 2021, Janzen and Hallwachs 2021, Weiss et al. 2024). Climate change is likewise increasing the frequency of other disturbances, such as fire and floods, both of which can have large-scale devastating effects on entire landscapes (Salcido et al. 2020, Dole et al. 2023, IPCC 2023). Increasing our understanding of how shifts in climatic variation, rather than shifts in mean conditions alone, influence long-term insect population trajectories remains a foremost research priority.

Disturbance: Fires and Floods

Shifting Fire Regimes Can Reshape Insect Communities and Accelerate Declines in Sensitive Taxa (Established but Incomplete)

Continued climate change and legacies of fire suppression have increased the frequency of conditions that promote fire, and the past two decades have seen the emergence of "megafires" (Jones et al. 2022, Linley et al. 2022). While low-intensity and high-frequency fires can have a positive impact on insect diversity and abundance (Burkle et al. 2019, Dell et al. 2019, Shafer et al. 2025, Nakas et al. 2026), catastrophic, large-scale fires are an emerging threat to global biodiversity, as their sheer extent can destroy the entire known range of some species while simultaneously reducing opportunities for recolonization (Dole et al. 2023, Latty and Forster 2025). For example, the 2019/2020 Australian megafires, which burned more than 30 million ha of land (Dickman 2021), burned the total known range of hundreds of invertebrate taxa (Marsh et al. 2022) and led to the death of an estimated 240 trillion arthropods (Dickman 2021). As a result, at least 11 bee species are now threatened with extinction under the IUCN guidelines (Dorey et al. 2021). Refugia can play a crucial role in post-fire recolonization (Gandhi et al. 2001, Adedoja et al. 2019, Bhaskar et al. 2019); however, the large extent of megafires can reduce the availability of such beneficial habitats, thus slowing or preventing recolonization and recovery. The impact of fire on insect populations is strongly mediated by life-history traits, ecological interactions, evolutionary history, and habitat (Koltz et al. 2018). For example, flying phytophagous and zoophagous arthropods often show greater resilience to fire, whereas species living in leaf litter, decaying wood, or the upper soil layers are less likely to survive or recover from burning (Moretti et al. 2006, Zaitsev et al. 2014).

Insect Responses to Floods Are Likely Highly Context-Specific (Inconclusive)

One of the primary consequences of climate change is the increase in extreme events, and the frequency of flooding has risen worldwide (Najibi and Devineni 2018, IPCC 2023). Flooding acts as a filter on both aquatic and terrestrial insect communities via direct and indirect pathways (Fies et al. 2016, Roeder et al. 2018, Romero et al. 2020, Santos et al. 2022).

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For example, the physical properties of a flood may directly induce mortality for some species and reduce population sizes (Fellendorf et al. 2004, McMullen and Lytle 2012), and major flooding events have been identified as a factor in the disappearance of entire genera of caterpillars in a tropical forest in Costa Rica (Salcido et al. 2020). Indirectly, flooding events can spread pollutants, pesticides, and other chemicals to new habitats (Davis et al. 2018, Zhou et al. 2022), thereby exacerbating other proximate drivers of insect declines. Many insects have some behavioral or physiological adaptations to cope with flood conditions; however, as floods become more stochastic, severe, and extensive due to climate change and anthropogenic interruption of hydrology (Huang et al. 2024), the deleterious effects of flooding are expected to increase. More research is needed to identify species and communities that are particularly susceptible to flooding and to quantify the characteristics of floods that drive insect declines. Restoring natural hydrological regimes, enhancing ecosystem resilience, and leveraging the biology and ecology of insects to inform flood mitigation strategies are expected to lessen the adverse impacts of flooding on both aquatic and terrestrial insects.

Pathogens

Pathogens Threaten the Health, Survival, and Reproduction of Their Hosts and Thereby Have the Potential to Contribute to Population Declines (Established but Incomplete)

Insects are naturally infected by a diverse range of pathogens, including bacteria, fungi, viruses, and protozoa. The capacity for pathogens to regulate insect population dynamics has been demonstrated, particularly among cyclic forest defoliators (Myers and Cory 2013) and outbreaking agricultural pests (Odindo 1983); however, relatively little is known of their role in driving population trends in other systems (Roy et al. 2009, Myers and Cory 2016). Pathogens have been established as key contributors to colony losses among managed honey bees (Brosi et al. 2017) and have also been proposed as potential drivers of population decline among wild bumble bees. For example, between June 2024 and March 2025, commercial beekeepers in the United States lost an average of 60% of their honey bees, with pathogens and parasites as a major contributing factor (Nearman et al. 2025). Likewise, elevated pathogen infection rates have been identified in North American bumble bee species experiencing declines, compared with relatively stable species (Cameron et al. 2011). The movement and spillover of pathogens from commercially reared and/or managed pollinators to wild insects has received considerable research attention (Fig. 3) (Meeus et al. 2011, Graystock et al. 2015, Fleites-Ayil et al. 2023), illustrating that agricultural practices can increase disease occurrence within wild insect communities (Fürst et al. 2014), with potential consequences for population stability (Schmid-Hempel et al. 2014). Historically, monitoring pathogen occurrence in natural insect populations has been rare and limited to well-studied hosts such as the western tent caterpillar (*Malacosoma californicum*) (Myers 2000) and the monarch butterfly (*Danaus plexippus*) (Altizer et al. 2000, Majewska et al. 2022); however, advances in molecular and genomic techniques have greatly facilitated the detection, quantification, and identification of entomopathogens and offer potential for further understanding the role of pathogens in contributing to population decline.

Combinations of Drivers

Insects Are Regularly Exposed to Multiple Anthropogenic Stressors (Well Established)

While drivers of insect decline are often presented as isolated and independent threats, the reality is that most threats are interconnected, and many populations face more than one threat simultaneously (Fig. 4) (Scherber 2015, Wagner et al. 2021, Wilson and Fox 2021, Yang et al. 2021, Halsch et al. 2025). Approximately a third of global surfaces have been transformed by land-use change since 1960 (Winkler et al. 2021), and the forces driving land-use change often introduce additional linked stressors that can further disrupt insect populations. For instance, agricultural expansion introduces additional chemicals for crop protection, including insecticides, fungicides, and herbicides, all of which can negatively impact non-target insects that are already suffering from habitat loss (Dudley and Alexander 2017). Likewise, urban environments introduce additional light, noise, and thermal pollution, all of which add stress to insects that have already lost habitat (Miles et al. 2019, Vaz et al. 2023). Some threats, such as air and water pollution, can have wide-ranging effects and impact environments far from their sources (Tilman et al. 2001, Beketov et al. 2013, Tang et al. 2021). For instance, pesticides applied by air or those that are water-soluble can move long distances and have been found in transported water and protected lands with no history of local pesticide applications (Halsch et al. 2020, Nowell et al. 2024). The stressor with perhaps the greatest interactive capacity is climate change, which poses an outsized threat because it influences nearly all ecosystems on Earth, thus impacting populations experiencing any subsets of other stressors (Staudt et al. 2013, Kaunisto et al. 2016, Scheffers et al. 2016). Taken together, insects are regularly exposed to multiple drivers of biodiversity loss, and understanding how this exposure translates into population- or metapopulation-level outcomes is one of the most pressing areas of research in insect conservation, especially when effects are often studied experimentally at the individual level.

Insects Experiencing Multiple Stressors Associated With Land-Use Change and Negative Effects Are Common (Well Established)

Studying combined effects at the population level is challenging because stressors are often highly correlated. Such work is far scarcer than that which focuses on a single effect, yet this area of research is rapidly growing. This has been better studied in the context of land use change, which often occurs alongside other threats and can be viewed as a progenitor of many additional stressors (González-Varo et al. 2013). One meta-analysis of the combined effects of land-use change and agriculture found that the most detrimental effects of intensive agriculture occurred in landscapes already simplified by land-use change (Batáry et al. 2011). Similar negative combinatory effects have been documented between land-use change and invasive species, in which invasive plants and pollinators are more abundant in modified landscapes (González-Varo et al. 2013) and between land-use change and pollution, where urban environments have greater pollution and fewer insects (Heinen-Kay et al. 2021, Vaz et al. 2023). Another widely recognized stressor of insect populations is pesticides, and it can be argued that pesticides are particularly likely to occur in combination with other stressors, as their use is almost exclusively on already modified lands. Interactive effects between nutrition and pesticide sensitivity are

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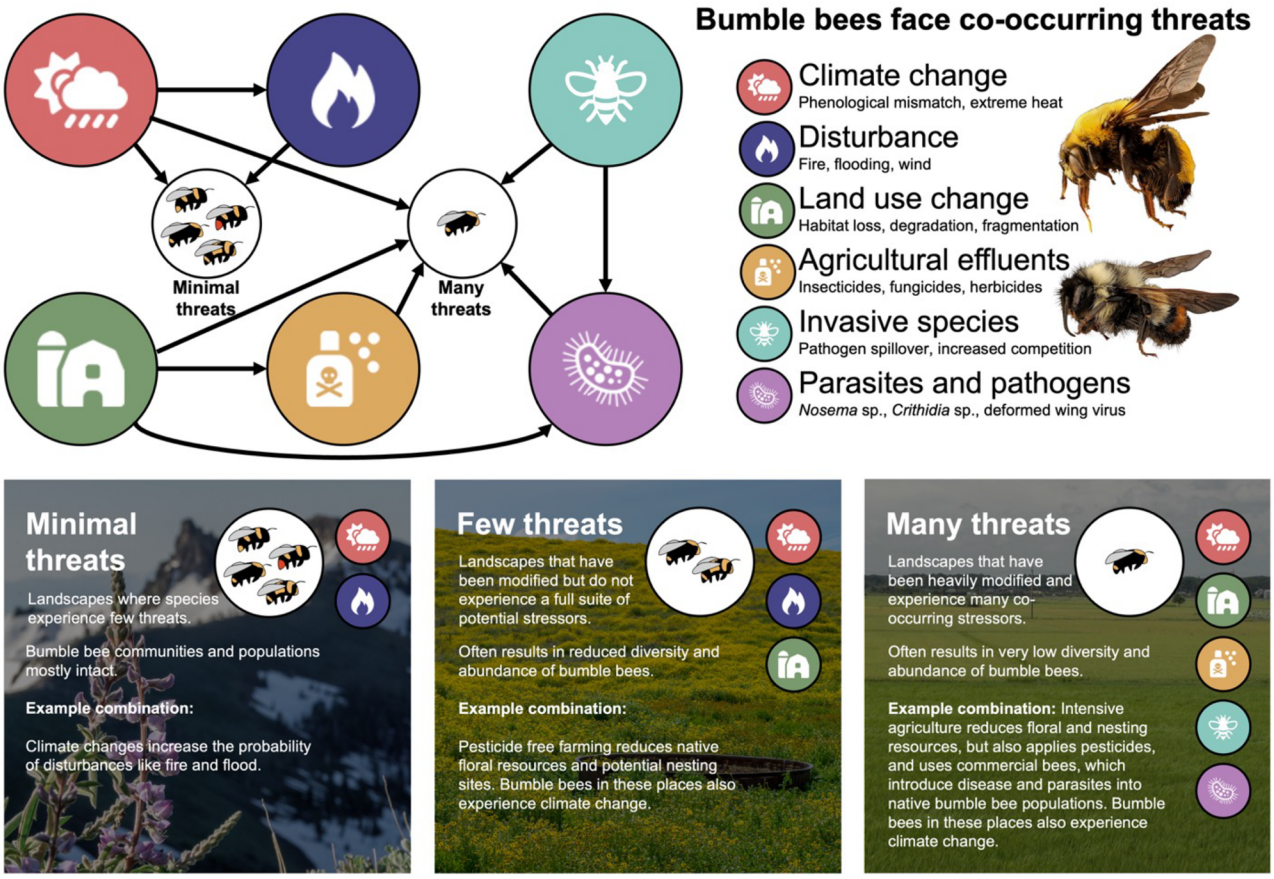


Fig. 4. The co-occurrence of anthropogenic stressors in bumble bees. In undisturbed landscapes, colonies may experience a limited number of stressors, perhaps only climate change. In contrast, in heavily modified landscapes, multiple threats can co-occur and interact. In agricultural systems, for example, bumble bee habitats become fragmented by changes in land use. Bumble bees are also exposed to pesticides in agricultural systems, can come into competition with managed bees (like honey bees) introduced for pollination, and have an elevated risk of parasites and pathogens, which can spill over from those same managed pollinators. Because these stressors are interconnected, they can accumulate rapidly within the same populations. Bee images courtesy of USGS bee lab.

well documented, and land-use change has greatly altered nutritional landscapes (Tissier et al. 2025). Further still, any driver such as land-use change that reduces insect population size or connectivity is likely to increase inbreeding and genetic drift, which in turn reduces standing genetic variation (Keller and Waller 2002, Frankham et al. 2017, MacDonald et al. 2025). Genetic erosion can therefore amplify the effects of habitat fragmentation, climate change, pesticides, pathogens, and other pressures. Where demographic and genetic diversity deteriorate together, initially external drivers can generate an extinction vortex in which reduced abundance, inbreeding, and loss of adaptive capacity become mutually reinforcing (Gilpin and Soulé 1986, Keller and Waller 2002).

Climate Change Has a Broad Capacity to Interact With Other Stressors (Established but Incomplete)

Interactions between climate change and land-use change have been documented in both agriculture and urban landscapes. One global study showed that sites with the greatest historical warming and the most intensive agriculture show reductions in species richness, an effect particularly strong in the tropics (Outhwaite et al. 2022), although other regional studies have found both negative and mixed effects on different species in response to both stressors (Fox et al. 2014, Ganuza et al. 2022). Warming temperatures interact with urbanization by exacerbating urban heat-island effects, which can have significant impacts, including shifts in phenology and reorganization of community structure (Diamond et al. 2015, Urban et al. 2024). Climate change also interacts with non-native species, with traits that make a non-native plant or animal more likely to establish, such as fast growth rates and high environmental tolerance, increasing their likelihood of responding positively to climate change (Schweiger et al. 2010). These processes can thus, together, entirely reshape native assemblages, with dramatic impacts on insect communities (Schweiger et al. 2010, Rafferty 2017). Temperature is also known to alter both the activity times and the physiological effects of pesticides on insects, and changes in temperature are likely to interact with both to alter pesticide exposure and underlying susceptibility (Albacete et al. 2023, Tissier et al. 2025). Finally, insect exposure and susceptibility to pathogens and parasites, which are already associated with modified landscapes (often containing non-native species and pesticides), can be modified by climatic factors (St. Leger 2021). For instance, the survival of the monarch butterfly (*D. plexippus*) decreases with extreme heat, but so does that of its parasites, highlighting the complexities of the conflicting effects of different stressors (Ragonese et al. 2024). Across these and the many other direct stressors

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highlighted in this section, the opportunities for interactions between climate change and each driver presented in this chapter are near limitless, and unraveling such synergistic and antagonistic effects is one of the most pressing areas of research on insect population pressures.

Indirect-to-Direct Drivers

While direct drivers should be seen as the immediate disruptive forces, indirect-to-direct drivers are the primary factors leading to the escalation of direct threats against insects. For instance, agriculture itself is often considered a threat to insects, yet it also causes other threats, such as land-use change and pesticide use.

Fisheries and Aquaculture

Industrial and Commercial-Scale Aquaculture Frequently Represents a Significant and Direct Threat to Insect Biodiversity (Established but Incomplete)

Aquaculture can incentivize the wholesale conversion of important insect habitats (e.g. mangrove swamps, estuarine backwaters, and floodplains) into fishponds and tanks (Diana 2009, Naylor et al. 2021, Rippel et al. 2021). This expansion has historic precedent (for example, dredging coastal swamps for fishponds was a regular feature of British colonial planning) and continues in drastic fashion today; for example, some coastal areas in India have seen multi-fold increases in just 3 decades as part of the so-called Blue Economy, resulting in the large reductions of wetland areas and their corresponding biodiversity (Jadhav 2024, Dar and Dar 2025). Aquaculture production (and associated fish waste) is also a major source of pharmaceuticals, heavy metals, plastics, and other pollutants and toxicants, whether released intentionally or accidentally, all of which can have lethal or sublethal consequences for aquatic invertebrates via reduced fitness, disrupted species assemblages, and altered ecologies (Namin et al. 2013, Minoo et al. 2016). Fish farming can also keep fish at unnaturally high densities, thereby introducing direct predation pressure on aquatic insects (Rantala et al. 2004, Šigutová et al. 2015). Fishing activity across all scales, but particularly at industrial levels, presents another direct threat to insect populations, as unsustainable levels of fish harvest can destabilize and disrupt freshwater ecosystems and food webs broadly, while some specific, if uncommon, fishing methods (e.g. cyanide or dynamite use) can cause broad harm to non-fish species, including insects (Kuparinen et al. 2016). Active stocking of lakes and rivers from fish hatcheries, a common practice to support recreational fishing economies, can also alter food webs in complex ways, with some insect species increasing in abundance while others become prey for introduced fish (Pope et al. 2009, Evangelista et al. 2025).

Agriculture and Grazing

Cropped Agriculture, Especially Its Intensification, Is a Primary Threat to Insect Biodiversity (Well Established)

The historic conversion of land for agricultural purposes is responsible for a 95% decline in wildlands from the 1700s to the 2000s (Ellis et al. 2010). The ensuing habitat loss has had incalculable effects on insect diversity; however, historically extensive agrarian systems are likely to have supported diverse insect communities, with such systems being seen as targets for conservation mitigation actions, e.g. European calcareous grasslands (van Swaay 2002, Hejcman et al. 2013). More recent intensification, characterized by crop breeding, more frequent harvesting, mechanization, and the use of pesticides, has led to monocultures and undermined the historic value of farmlands (Clar and Pinilla 2011, Raven and Wagner 2021). Large-scale mechanization has driven an increase in field sizes, with a concomitant decrease in critical small-scale, semi-natural habitat features, and habitat connectivity important for insect populations (Haddaway et al. 2018, Raven and Wagner 2021, Priyadarshana et al. 2024). Mechanization has also introduced new cultivation techniques that can affect soil communities (Postma-Blaauw et al. 2010), while inorganic fertilizers permit more frequent sward cuts in grassland systems, including silage-based forage systems (Frame 2000, Humbert et al. 2010). The widespread use of pesticides and inorganic fertilizers poses additional challenges, as all insecticides and many fungicides directly target insect physiology (Serrão et al. 2022, Wolfram et al. 2026). Fertilizers and herbicides can further modify plant nutrient content and plant communities, which carry indirect consequences for insects. The immediate impacts of these chemicals are felt in agricultural lands, but wider, diffuse pollution also affects adjacent natural and semi-natural habitats that may be critical for disrupting insect populations (Whatley et al. 2015). Recently, there have been significant shifts in attitudes toward sustainable agricultural systems, and there is ongoing debate about the extent to which demands for global food security complement or conflict with the need to support insect biodiversity, which provides many services important for agriculture (Jachertz and Nützenadel 2011, Pretty 2018, Pecenka et al. 2021, Wyckhuys et al. 2023, Woodcock et al. 2025a).

Pasturelands Comprise the Largest Fraction of Agricultural Lands Globally, and Their Impacts on Insects Are More Variable Than Those of Cropped Agriculture (Established but Incomplete)

While cropped systems have received the bulk of attention, the largest share of agricultural land is used for livestock, which currently comprises 30% of ice-free land (Dangal et al. 2017). Heavy grazing can fundamentally reshape landscapes, from the soil to plant communities, and these landscapes are often associated with widespread invasion of non-native plants (Alkemade et al. 2013, Jauni et al. 2015, Lai and Kumar 2020). At the same time, these impacts vary with the environmental context and management, and grazing has also been used for conservation and to prevent further degradation (Frost and Launchbaugh 2003, Franzluebbers et al. 2012). Livestock grazing has been shown to disrupt nutrients in streams, which are associated with reductions in the richness of aquatic insects, although neutral effects have also been observed (Strand and Merritt 1999, Barnes et al. 2025). The effects of ranching on terrestrial insects are less well studied, but negative associations are documented between livestock grazing and Coleoptera, Diptera, Hymenoptera, and Orthoptera diversity, whereas Hemiptera have been found to be more diverse in grazed sites in the southeastern United States (Kruess and Tscharntke 2002, Debano 2006, Steck et al. 2007, Le Gall et al. 2019). At the same time, the stocking of sheep and livestock has been used to restore habitats of European butterflies, and grazing is often used as a tool for ecological restoration (Pöyry et al. 2004, WallisDeVries et al. 2016, Bussan and Schultz 2023). Relative

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to croplands, pasturelands are not associated with the same level of chemical contamination (with notable exceptions; see Martin et al. 2022) and can be botanically diverse, and, as a result, pose a lower overall threat to insect biodiversity than crops. Given the large area that pasturelands cover globally and their relatively unconverted state (compared with intensive croplands or urban lands), further investigation of their impacts and potential benefits for insects remains important.

Forestry

Forestry Practices Significantly Reshape Forest Structure and Resources, With Mixed but Often Negative Impacts on Insect Communities (Well Established)

A large proportion of global forests is managed, and forestry thus influences the rich diversity of insect species that depend on these ecosystems (FAO 2020). All forms of forest management will alter forest properties and may, depending on the species, have both positive and negative effects on insect populations. For instance, large-scale tree removal reduces the availability of key resources such as wood and foliage, lowers structural integrity, and alters microclimates. Forest management efforts tend to favor practices that maximize yield, resulting in even-aged stands, whereas insect diversity is favored by structurally complex forests (Knuff et al. 2020). Many forest-adapted insects have narrow thermal ranges and often cannot cope with the hotter, more variable microclimate in logged or cleared forests (Williamson et al. 2022). Forestry often reduces both standing and fallen deadwood availability by removing trees before they die naturally. Thus, managed forests frequently have a deficit (Müller and Bütler 2010) of this crucial resource, on which many forest insects, especially saproxylic beetles and flies, depend (Stokland et al. 2012). Through forestry, non-native tree species have been propagated worldwide and are mostly planted in monocultures. As tree diversity is a key determinant of insect diversity (Lombardi and Mali 2016, Ulyshen and Šobotník 2018, Schuldt et al. 2019), such monocultures reduce insect diversity, although a global meta-analysis found evidence of this effect only for predators and parasitoids, not for herbivorous insects (Wan et al. 2020). Most knowledge of the impacts of forestry on insects comes from temperate forests, and there is a great need for parallel studies in tropical forest systems.

Harvesting

The Benefits to Humans of Harvesting Insects May Outweigh the Costs to Focal Species (Inconclusive)

Insects are collected for a variety of uses, including research, education, trade, and entomophagy. Insects differ from vertebrates in that they have short generation times (a year or less for most species), a prodigious capacity for reproduction, and populations that may number in the millions. As a result, many insect populations can recover rapidly from losses, although those with lower growth rates would be expected to be more susceptible to harvesting (Bauer et al. 2013). In some specific cases, over-collection of particular species for trade, including endangered insects, especially those considered aesthetically valuable, does occur (Wang et al. 2023) and can locally deplete populations (Putri 2016). However, given that relatively few species are harvested, this activity is not considered a globally significant threat (Pohl 2009). At the same time, insect collections often provide vital specimens for taxonomy and biodiversity research, as well as for education and outreach, and serve as entry points for professional entomologists (Kharouba et al. 2018, Davis et al. 2023). Overall, the conservation benefits of insect collecting, in terms of knowledge gained and public interest generated, may outweigh the ecological costs, although data to support this statement are largely lacking.

Mining

Mining Activities Can Have Significant Local and Regional Impacts on Ecosystems (Established but Incomplete)

Mining can have long-lasting effects on ecosystems, from the initial development and extraction to the long-term legacies of pollution, water extraction, and land-use change (Carvalho 2017). The effects of mining on insects vary by context but are often adverse (Harabiš 2016, Hughes 2017). Key effects include habitat fragmentation, pollution of waterways, and microclimate changes, such as exposed ground altering reflectance and heating under direct sunlight (Boldy et al. 2021, da Silva-Rêgo et al. 2022). These pressures can decimate specialist species while sometimes favoring disturbance-tolerant generalists (Stoll et al. 2022, Rivera-Pérez et al. 2024, Silva et al. 2025). Additionally, the support infrastructure associated with mining, such as roads, water demands, processing facilities, settling ponds, and camps, continually reshapes forest landscapes worldwide, altering insect communities and, in turn, impacting critical ecosystem services (Sonter et al. 2017, Boldy et al. 2021). These effects may be especially significant in developing countries, where relaxed land-use regulations facilitate new clearings (Thomashausen et al. 2018).

Infrastructure

Infrastructure Is Generally Thought to Have Adverse Effects on Insects, but the Extent of Its Impact Varies by Type (Established but Incomplete)

The presence of infrastructure, including roads, railways, power lines, pipelines, and dams, has risen dramatically with human population growth. Generally, infrastructure causes physical and/or functional habitat loss, acts as a physical or behavioral barrier, creates edge effects, and degrades surrounding habitat quality through pollution, including air, noise, light, water, thermal, electromagnetic, and soil pollution (Spellerberg 1998, Jones et al. 2015, Villemey et al. 2018). Among terrestrial infrastructures, roads have been studied most extensively. Road traffic kills billions of insects annually (Baxter-Gilbert et al. 2015), but the short- and long-term effects on insect populations remain poorly understood. Management of vegetation along road verges can have direct (e.g. mowing, pesticides) and indirect effects (through changes in vegetation) on insects (Jakobsson et al. 2018). Some of these effects are independent of traffic volume, but habitat loss, barrier effects, and collision-related mortality increase with traffic. Dams represent another form of infrastructure modification to natural systems and are among the most widespread disrupters of river systems worldwide. Dams significantly alter the community structure of aquatic macroinvertebrates, primarily by decreasing diversity through downstream substrate modification (Van Looy et al. 2014, Wang et al. 2020). These impacts are highly context-dependent and vary with dam size and location, both within the watershed and with climate and elevation (Wang et al. 2020).

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Tourism

Direct Evidence for Tourism-Related Impacts on Insects Is Weak to Moderate and Focused on Specific Ecosystems, Tourism Activities, and Insect Taxa (Inconclusive)

Hiking and outdoor tourism can locally diminish terrestrial and aquatic insect populations and alter insect services (Escarpinati et al. 2014, Ciach et al. 2017, Kamel 2020, Pablo-Cea et al. 2021), although one study in British woodlands also showed no measurable effects of tourism hiking on forest insect communities (Kennett et al. 2024). Tourism activities in beach-dune ecosystems can suppress insect communities either directly (e.g. via trampling, off-road vehicles) or indirectly through habitat modification (e.g. via mechanical beach cleaning), varying with the intensity of tourism pressure (Arndt et al. 2005, Comor et al. 2008, Fantinato 2019, Zanella and Vianello 2025). Cave-dwelling insects can be negatively affected by tourist activities in some cases (Pacheco et al. 2021) but not in others, and this is probably related to the intensity and management of tourism (Faille et al. 2015). Indirect consequences of tourism can also be seen in pesticide applications around tourist resorts, which have been linked to insect declines and alterations in food webs on tropical islands (Steibl et al. 2021, 2022). Tourism-related travel also contributes to the global spread of non-native, invasive insect species, particularly to island ecosystems and regions lacking adequate biosecurity measures (Liebhold et al. 2006, Roques 2010, Bertelsmeier et al. 2025). Alternatively, entomo-tourism, the promotion of insect diversity as a wildlife experience or of particular populations of focal species, such as the synchronous firefly (*Photinus carolinus*) or monarch butterflies (Lemelin and Jaramillo-López 2020), is a nascent concept in ecotourism (Kanaujia et al. 2022); however, there is currently limited direct evidence supporting its effectiveness as a conservation tool (Lewis et al. 2021, Guerrero-Moreno and Oliveira-Junior 2024).

Relocation of Goods and People

Movement of Goods and People Facilitates the Introduction of Harmful Organisms (Well Established)

Human movement has shaped insect distributions across history. In ancient times, mass migrations carried insect parasites, and contemporarily, they have spread around the world through immigration, displacement, labor migration, and trade (Steverding 2020). The widespread adoption of agriculture, European colonization, and modern globalization have also moved plants and insects around the globe (Miedaner and Garbelotto 2024). The consequences of such movements are demonstrated by honey bee species (*Apis* spp.), which have been introduced across the New World for pollination services and, in the process, have introduced parasites and increased competition with native insect fauna (Meeus et al. 2011, Bertelsmeier 2021). Travel and movement of commodity items for commerce are also key ways in which insect species move into new areas via human-mediated dispersal (Gippet et al. 2019). Global transportation and trade networks have introduced over 6,700 insect species into new areas (Bertelsmeier et al. 2025) and contributed to the spread of insect vectors of infectious disease (Soto 2009).

Indirect Drivers

Indirect drivers of change are the human factors that lead to direct impacts on insect biodiversity. For instance, poor human perception of insects does not do direct harm, yet it can have detrimental effects on insect conservation. These categories have been directly adapted from the IPBES global assessment report, and, where possible, examples relevant to insects have been provided; however, given the nature of indirect forces, this was not always possible. In these cases, the links between indirect force and the general erosion of nature are presented, which are undoubtedly relevant to insects.

Values

Limited Awareness of Insect Diversity and Services Diminishes Their Perceived Value (Established but Incomplete)

Insects suffer from a deficit of positive public interest compared to larger-bodied vertebrates in both social and scientific arenas (Fig. 3) (Troudet et al. 2017, Samways et al. 2020). Empathy and compassion decrease with phylogenetic distance from humans (Miralles et al. 2019), which may hamper interest in, knowledge of, and ultimately the perceived value of insects. As a consequence, many insects are under-represented in conservation science (Clark and May 2002, Leather 2009) and are less likely to receive conservation funding (Leather 2009). Further, insects are less prominent in public-facing conservation narratives than vertebrates (Robinson 1991, Guiney and Oberhauser 2009, Rose et al. 2019, Brereton and Brereton 2020, Howlett et al. 2023, Mammola et al. 2023), and people may be disinclined to support insect conservation due to their status or perceived status as human or non-human animal parasites, "pests" of crops and other plants, or vectors of disease (Wise and Borry 2022, Barrett and Fischer 2023). Cases of human–wildlife conflict reflect trade-offs among our values, where the supposed "cost" of insects (financially, to human health, etc.) is at odds with the value of the insects themselves. It is likely that such trade-offs are unintentional, in which the ecological cost is not at the forefront compared to other aesthetic, economic, or sociocultural values. Additionally, some insect species may lack perceived beauty or aesthetic value, or be devalued based on superficial traits, leading larger, more attractive insects to be more likely to receive legal protection (Leandro et al. 2017).

Improved Perception May Contribute to the Conservation of Insects and Foster Attitudes of Preservation (Established but Incomplete)

In some contexts, insects are valued for aesthetic, economic, cultural, and ecosystem service reasons, which can influence conservation sentiments (Losey and Vaughan 2006, Morimoto 2020, Duffus et al. 2021). Insects may be valued non-specifically, as one part of a broader "ecological stewardship" mindset, particularly if people care about the intergenerational value of thriving ecosystems. Many papers have argued that emphasizing the ecosystem or societal benefits of insects is key to increasing their social valuation (Didham et al. 2020, Morimoto 2020). Further research is needed to understand the extent to which "social benefit" contributes to a positive insect conservation mindset. It has been suggested that incorporating insects into cost-benefit analyses may elevate their perceived importance; however, economic valuation of nature is itself controversial, and economic arguments may not foster conservation attitudes as much as appealing to other values (Bolderdijk et al. 2013, Byerly et al. 2018). Increasing knowledge of and

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interaction with insects would be expected to reduce innate negative attitudes toward them, though the impacts of this improved perception of conservation actions warrant further study (Chalmin-Pui and Perkins 2017, Fukano and Soga 2023, Gish et al. 2024, Leto et al. 2024).

Demographic

Human Population Growth Continues Globally, With Rates Varying by Region (Well Established)

The global population is projected to continue growing for decades, potentially reaching over 10 billion by 2080 (United Nations, Department of Economic and Social Affairs, Population Division 2024). While populations and resource consumption have increased, biodiversity has declined, and these factors are significant drivers of change in the biosphere (Crist et al. 2017). More than just overpopulation, inequality in resource consumption is a key driver of biodiversity loss, as higher-income economies exert pressure on smaller, and often more biodiverse countries (Hughes et al. 2023). Over 1.1 billion people across 109 countries live in multidimensional poverty (UNDP [United Nations Development Programme] 2025), and regions of greater wealth have lower levels of biodiversity (Kubiszewski et al. 2024). The world population is also expected to get older, with the median age increasing and the number of people aged over 65 predicted to reach 1.5 billion in 2050 (United Nations, Department of Economic and Social Affairs, Population Division 2001). Because of this, many countries already have declining populations, which, combined with migration, will reshape the distribution of human population density (United Nations, Department of Economic and Social Affairs, Population Division 2024). While demographic change is associated with decreased biodiversity, the relationship is complex and confounded by uneven spatial patterns and geopolitical factors (Mehring et al. 2020).

Demographic Change Is an Upstream Cause of Many Known Drivers of Insect Biodiversity Loss (Well Established)

Residential areas are expanding faster than populations are growing, in some cases near biodiversity hotspots (Seto et al. 2013). Overall, the conversion of land for urban use has been observed to have mixed impacts on the diversity and abundance of terrestrial arthropods; however, these impacts depend on what land type is being replaced (Fenoglio et al. 2021). Human migration has impacted biodiversity globally over centuries (Oglethorpe et al. 2007) and has led to the introduction of both invasive species and pathogens (Perrings et al. 2009).

Technological

New Technologies Have Increased the Footprint of Some Drivers but Offer Promise for Biodiversity Monitoring (Well Established)

Technological improvement has increased the scale and intensity of land conversion and degradation. Technological effects on insects are well demonstrated in agricultural systems, where mechanization has led to simplified landscapes that are unsuitable for maintaining high insect diversity (Raven and Wagner 2021). In the same landscapes, the development of water-soluble pesticides has increased contamination of natural areas, a phenomenon well demonstrated by neonicotinoids (Main et al. 2018). At the same time, improvements in agricultural technology, in theory, can reduce pesticides and fertilizer drift and lessen the need for further deforestation; however, forest loss for agrarian use continues, especially in the tropics (Grainger et al. 2003, Pendrill et al. 2022). While some technological advances have exacerbated land-use change, advances in monitoring and molecular methodologies have greatly advanced biosystematic studies of insects. Technologies such as visual and acoustic monitoring, barcoding, eDNA, and next-generation sequencing are providing new data and insight into fundamental ecological and evolutionary processes and offer an excellent opportunity to understand changes in insect biodiversity, particularly in tropical regions (Hebert et al. 2003, van Klink et al. 2022, Høye and Grames 2025, Riede and Balakrishnan 2025, Grele and Richards 2026).

Economic

Social and Economic Trends Impact Species and Ecosystems (Well Established)

While socioeconomic trends impact all biodiversity (Liu et al. 2003), certain trends have a disproportionate effect on insects. For instance, following the conclusion of World War II, small and diversified farming practices in temperate regions gave way to industrial agriculture, characterized by both intensive land-use change and the addition of pesticides formulated to target insects (Robinson and Sutherland 2002, Raven and Wagner 2021). More recently, land-use change for agriculture has spread to more biodiverse tropical regions, posing a significant threat to the large pool of insect diversity in these areas, much of which remains undescribed (Green et al. 2019, Boyle et al. 2025). These changes are driven by global pressures for continued food security; for example, in Brazil, deforested land is used for beef exports (zu Ermgassen et al. 2020). Climate change, whose effects were previously described, is a direct consequence of greenhouse gas (GHG) emissions. Emissions continue to rise globally, and while some countries have reduced total per capita emissions, concern persists that the growing agricultural influence in tropical countries will further accelerate GHG emissions (Lamb et al. 2021). While agriculture is a major contributor to GHG emissions in tropical regions, globally, energy and industry remain the largest contributors.

Market Interactions

Market Forces Influence Global Patterns of Ecosystem Disruption (Well Established)

Researchers increasingly recognize that market shocks and policy changes in one country or jurisdiction can have local and global impacts on natural systems (Carrasco et al. 2017, Hull and Liu 2018); however, methods for attributing and measuring the strength of these relationships are still emerging (Tonini and Liu 2017). Several recent examples of linkages between markets and land-use decisions illustrate how economic policies and market shocks can influence land-use change. When Chinese tariffs on soybean imports from the United States were instituted in 2018, Brazilian growers expanded soybean acreage by 4 million ha, converting 67,000 ha of natural land cover in the process (Richards et al. 2020). Econometric research has identified a robust causal relationship between global palm oil prices and deforestation rates across Indonesia (Cisneros et al. 2021). At the national scale, the US government strengthened policies boosting demand for ethanol biofuels beginning in 2007, which resulted in a dramatic decline in crop diversification across the American

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Midwest as growers increased production of corn by 9.7% and soybeans by 6.7% at the expense of other crops, contributing to landscape simplification (Smith and Moschini 2024).

Integrative Summary

Reductions in biodiversity are apparent across insects (Wagner 2020). Compared with larger-bodied organisms like birds and mammals, insects as a group have less data and more volatile population dynamics, which increases uncertainty in forecasting individual, population, and community responses (Bahlai 2026). While these dramatic population responses pose challenges for prediction, they also illustrate the remarkable potential for insects to recover and adapt to rapidly changing environments.

Despite issues with data availability, the major threats facing insects are well established and broadly align with those affecting other taxa. There are, however, particular threats, such as insecticides or light pollution, that pose a disproportionate risk to insects compared to other taxa. Many threats act at highly localized scales and can have dire consequences in those situations, such as invasive animals or tourism in island systems; others, such as land-use change, including habitat loss and degradation, along with climate change, are jeopardizing populations, species, and communities across the world. We know relatively little about how these global stressors, acting together with local ones, will disrupt insect populations in additive and synergistic ways. Interactions among different drivers are likely critical for predicting responses and prioritizing conservation, and resolving such effects should be a primary goal of insect conservation research.

A further challenge is that current knowledge is unevenly distributed across taxa and geographic regions. Most evidence on the drivers of insect declines is based on a few relatively well-studied insect groups. For example, we know more about how many drivers affect pollinators than we do about other groups, but we lack knowledge of the extent to which these effects are generalizable to other insect groups. Ecological responses are also far less well documented in tropical than in temperate regions, even though some of the most rapidly changing landscapes on Earth occur in the tropics. This is particularly troubling given that insect diversity is far greater at low latitudes, and we can expect extinction to occur faster than new species will be described, posing a challenge to comprehending rates of loss. While uncertainty remains about insect responses to anthropogenic pressures, sustained interest over recent decades has yielded substantial insights, and further research will continue to refine our knowledge and improve conservation efforts. A lack of a comprehensive understanding should not impede immediate action (Forister et al. 2019, Grames 2026, Elphick et al. in this issue).

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