Connecting people, connecting landscapes

Edited by
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Manisha Bhardwaj, Denis François, Ivo Dostál

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Editorial

Mainstreaming biodiversity into transport networks by connecting stakeholders across sectors

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Linear Transportation Infrastructure drives global biodiversity loss

Habitat fragmentation and loss are considered the main causes of global biodiversity decline (Barnosky et al. 2011; Hilty et al. 2020) and have received increased attention in recent decades (MacArthur and Wilson 1967; Wilcove et al. 1986; Saunders et al. 1991; Fahrig 2003; Henle et al. 2004; Fletcher et al. 2018). Among the many causes of fragmentation and habitat loss, development of linear transportation infrastructure (LTI), particularly roads, railways and energy infrastructure are the main drivers (Seiler 2003; Geneletti 2004; Rhodes et al. 2014). While LTI plays an important role in providing mobility and connectivity for people, infrastructure occupies land, can disrupt natural processes, facilitates the spread of invasive species and imposes movement barriers to most terrestrial wildlife (Seiler 2003, 2014, 2023; Smith et al. 2015; Soanes et al. 2024). Furthermore, the negative impacts of LTI on the environment, quickly accumulate and spread beyond the site of infrastructure, through, e.g., traffic mortality, noise disturbance, and chemical pollution (Forman and Alexander 1998; van der Ree et al. 2015; Hlaváč et al. 2019; Denneboom et al. 2024). With the planned expansions of LTI networks and expected increase in traffic, the cumulative impact on nature will quickly exceed the carrying capacities of ecosystems (Forman and Alexander 1998; Jaeger and Torres 2021).

In recent decades, scientific literature on understanding and mitigating the negative impacts of LTI has significantly grown (van der Ree et al. 2015; Seiler and Bhardwaj 2020; Sjölund et al. 2022; Barnot et al. 2023). To date, most literature and policies (e.g., Natura 2000; EU Green Infrastructure Strategy) agree that habitat fragmentation effects are best resolved through improving landscape permeability, by provisioning structural connectivity, through, e.g., ecological corridors as part of coherent ecological networks (Loro et al. 2015; Mimet et al. 2016; Vlkova et al. 2024). When ecological corridors successfully facilitate the movement of wildlife, and genes flow through the landscape, corridors are deemed to provide "functional connectivity". Despite the wide array of solutions and tools that have been developed to improve structural and functional connectivity in the landscape, sufficient adoption of such solutions is missing, and functional connectivity is not effectively facilitated on the landscape-scale. It is evident that sustainable solutions require stronger transdisciplinary cooperation among stakeholders and specialists, such as transport administrations and development financiers, as well as the general public (Papp et al. 2022a), to foster the development of clear standards and procedures that ensure the effectiveness of biodiversity conservation efforts while developing transport networks (Papp et al. 2022b).

Infrastructure and Ecology Network Europe in the framework of transport ecology

The Infrastructure and Ecology Network Europe (IENE) has pioneered a transdisciplinary collaborative approach in the LTI sector. Established in 1996, IENE provides an independent, international and interdisciplinary platform for developing and exchanging expert knowledge, with the aim of promoting a safe, meaningful and ecologically sustainable pan-European transport infrastructure (https://iene.info/). The network brings together decision makers, institutions responsible for LTI planning and development, environmental protection agencies, researchers, academia, practitioners, consultants, businesses and relevant NGOs. IENE facilitates dialogue and collaboration between all these key stakeholders, through various initiatives and events, including its biennial international conferences. One such conference, ”Connecting people, connecting landscapes” was organised by IENE in September 2022, in Cluj-Napoca, Romania. The conference aimed at finding integrated approaches to mainstream biodiversity into transportation networks by assessing the current state of play, discussing the gaps, needs and solutions, looking back for lessons learned and ahead for future challenges and opportunities, from global, European and regional (Carpathians, Danube, South East Europe and Black Sea) perspectives. The main themes included: (1) Mainstreaming biodiversity into the transport sector (including infrastructure and energy networks); (2) Practical experiences, challenges and opportunities related to transport ecology and (3) Integrated solutions for ecological connectivity. The conference attracted 276 participants from 46 countries across Europe, Asia, Africa, Australia, and the Americas, who exchanged knowledge and expertise over 190 oral presentations, workshops and panel discussions. These sessions addressed a wide range of topics, including sectoral policies, financing, strategic planning for LTI, environmental impact assessments, design, implementation, operation, upgrading, and decommissioning of LTI, as well as monitoring, research, communication, awareness-raising, education, and fostering effective consultations and collaborations. In addition, four thematic field trips were organised for in-person attendees, offering first-hand insights into the challenges and opportunities posed by both green and grey infrastructures. These trips deepened participants’ understanding of how infrastructure and biodiversity can coexist and highlighted innovative approaches to overcoming the practical challenges of LTI planning and implementation.

About this special issue

This special issue, titled “*Connecting People, Connecting Landscapes*,” features selected research and case studies presented during the IENE 2022 International Conference. The issue consists of 8 papers covering Europe (6), North America (1), and Asia (1); focusing on various infrastructures, including roads (6), railways (3), roads and railways combined (2), waterways and power lines (1). The key topics addressed by these papers include wildlife crossings (3), land use near wildlife crossings (2), ecological connectivity (2), environmental impact assessments and mitigation measures for LTI (2), prevention of animal-vehicle collisions (2), road fencing and electrified barriers (1), and the role of LTI as wildlife habitat and refuge (1).

Wildlife crossings can facilitate animal movement across landscapes and mitigate human-wildlife conflicts, particularly for LTI. Maierdiyali et al. (2024) provide a comprehensive study from the Tibetan Plateau, China, examining factors that influence wildlife use of underpasses along highways, expressways, and railways. They find that the use of underpasses is strongly correlated with their size and location, with larger and more isolated underpasses being preferred by the species studied. Similarly, Jurečka et al. (2024) conducted research in Austria on wildlife crossing structures at the intersection of ecological corridors and road infrastructure. Their study analysed both the usage of wildlife crossings and species richness in relation to land use and human activity. They found that wildlife overpasses are the most effective, but individual characteristics play a critical role in their success. Furthermore, mammal species richness was positively associated with higher vegetation cover and reduced human presence and disturbance. These studies highlight the importance of strategically planning wildlife crossing structures along key ecological corridors while minimising intensive land use and high human activity to maximise their effectiveness.

The appropriate fencing of LTI to reduce wildlife-vehicle collisions (WVC) remains a key focus in road ecology. A study in Montana, USA, investigated the use of electrified barriers to deter black bears (*Ursus americanus*) from entering fenced roads, specifically at low-volume access points, such as side roads and driveways leading to agricultural fields (Huijser and Getty 2024). Conducted on private land at a melon patch—a known attractant for bears—the researchers found that fences with well-designed, operated, maintained, and monitored electrified barriers successfully kept almost all black bears out of the melon patch, effectively breaking their habitual foraging behaviour. These electrified barriers prove especially crucial along road sections where wildlife fences need to exclude species with paws, such as bears, from entering fenced road corridors.

The integration of predictive models and Artificial Intelligence in preventing WVC is a rapidly growing field. Moulherat et al. (2024) developed a pioneering framework in south-western France aimed at managing WVC by mapping collision risks between trains and ungulates, especially roe deer (*Capreolus capreolus*) and wild boar (*Sus scrofa*), using a network of camera traps. This framework utilised population dynamic simulations to pinpoint collision hotspots and optimise sensor deployment strategies. Data collected from camera traps was processed with deep learning algorithms to detect and identify species near LTI. The study highlighted the technical and operational requirements necessary to effectively integrate biodiversity concerns into LTI digital twins. This advancement has significant potential for reducing WVC by enabling dynamic, adaptive mapping systems that could provide real-time alerts to connected (and even autonomous) vehicles across various transport infrastructures.

Wildlife-vehicle collisions involving threatened species pose a significant conservation challenge. Niemi et al. (2024) conducted an analysis of traffic-related mortality of the endemic European wild forest reindeer (*Rangifer tarandus fennicus*) in Finland between 2017 and 2022. The study recorded 259 reindeer killed in road traffic collisions or euthanized later after tracking, and at least 52 individuals killed following railway incidents. Interestingly, adult reindeer were more frequently involved in collisions than juveniles, with nearly equal representation of adult males and females. These findings highlight the urgent need for species-specific mitigation strategies, such as identifying collision hotspots and deploying wildlife detection systems and warning signs. Such measures would not only protect endangered wildlife but also enhance human safety on roads.

Preserving landscape connectivity during the development of new LTI is critical for maintaining ecological processes. Domokos et al. (2024) conducted a study assessing bear presence and genetic connectivity across a proposed highway in the Eastern Romanian Carpathians, while also estimating the minimum population of brown bears (*Ursus arctos*) in the area. The study identified functional ecological connectivity across the planned highway, demonstrated by genetic links between the 24 sampled bears. Bears frequently appeared near the proposed highway, especially in rugged terrain, and were often detected close to human settlements (<1 km). Even before highway construction, connectivity appears limited by an extensive network of settlements, leaving only a few key linkage areas undeveloped. With wildlife crossing structures inadequately planned for this highway, the authors recommend conducting permeability studies post-construction to preserve landscape connectivity, as the highway could otherwise severely disrupt the Romanian and broader Carpathian bear populations.

The impact of new and planned transport infrastructure on biodiversity and socio-economic systems is widely recognized, yet the effects of ageing infrastructure on nature and human society are often overlooked. Dostál et al. (2024) developed and tested a methodological framework in the Czech Republic aimed at addressing environmental issues associated with older transport infrastructure. The framework presents a systematic approach for the preventive identification of problematic hotspots on existing road networks, proposing feasible upgrades or optimizations that can be integrated into routine repairs and small-scale reconstructions. It outlines a process that includes preparation of assessment backgrounds, field survey protocols, and the design and monitoring of mitigation measures. Fourteen key environmental problem areas were identified. The framework's comprehensive methodology has strong potential for application in other countries, including post-project evaluations of newly constructed roads.

François et al. (2024) analysed the role of LTI rights-of-way of roads, railways, waterways and power lines, as an ecological shelter for biodiversity in France. They developed a GIS-based methodology to estimate the linear extent and surface area of these potential ecological shelters, with a focus on local flora and entomofauna. Their goal was to propose and optimise policy actions that could enhance the role of LTIs in providing sustainable habitats. The study suggests that implementing targeted management strategies for these areas could enhance their function as refuges for local wildlife and even serve as source populations for recolonizing adjacent degraded landscapes, thus creating broader ecological benefits. Achieving this requires active participation from a wide range of stakeholders, including state authorities, LTI operators, and local landowners. In some cases, new responsibilities may need to be assigned to ensure effective management. Such an approach not only benefits protected wildlife but also supports common species, which are often overlooked despite their critical ecological roles and functions.

Perspectives

The IENE network holds significant knowledge, experience, and best practices with the potential to effectively integrate biodiversity into transport networks. The IENE 2022 International Conference served as an ideal platform for exploring how to achieve this integration, bringing together a variety of stakeholders, including policymakers, transport and environmental agencies, researchers, academics, and NGOs. The outcomes of the conference proceedings, as well as the findings of various studies, such as those presented in this Special Issue, provide valuable insights that can guide both policy and societal transformations. To ensure success, transdisciplinary collaboration must be encouraged, and stakeholder participation and co-creation should be prioritised from local to international levels. By fostering connections among people, we can create the conditions necessary to preserve landscape connectivity, benefiting both human and natural ecosystems.

Additional information

Conflict of interest

The authors have declared that no competing interests exist.

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Papp CR designed the manuscript and developed the first draft, the other authors contributed with input and feedback. Papp finalised the article.

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Data availability

All of the data that support the findings of this study are available in the main text.

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Editorial

Connecting people to connect landscapes

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Linear transport infrastructure (LTI) networks—including roads, railways, navigation canals, irrigation systems, and power lines—are vital for socio-economic development, human convenience, and overall prosperity (Srinivasu and Rao 2013; Skorobogatova and Kuzmina-Merlino 2017). However, much of this infrastructure, particularly in the last few decades, has been constructed with little regard for its adverse effects on biodiversity and wildlife movement (van der Ree et al. 2015; Torres et al. 2016; Bennett 2017). Additionally, the cumulative impact of multiple infrastructure networks at the landscape level, combined with other man-made and natural barriers, has often been overlooked (Papp et al. 2022a).

Although the approach to LTI development has evolved as the shortcomings of earlier practices became evident, significant challenges remain in effectively integrating biodiversity considerations into both the modernization of existing infrastructure and the planning of new projects (Hlaváč et al. 2019; Rosell et al. 2023). Key obstacles include a lack of strong cooperation between stakeholders, insufficient transparency, weak stakeholder engagement, and a general lack of motivation to address biodiversity concerns. Overcoming these challenges requires fostering transdisciplinarity (Papp et al. 2022b), including a culture of collaboration, ensuring meaningful participation, and creating incentives for sustainable infrastructure development that integrates biodiversity and ecosystem services.

The rapid advancement of technology and the push for faster, safer transportation systems often outpaces the ability of transdisciplinary approaches to address other critical concerns, such as wildlife movement across LTI (Seiler and Helldin 2006; Seiler and Bhardwaj 2020). The Infrastructure Ecology Network Europe (IENE) (https://www.iene.info/), founded in 1996, has been a pioneer in promoting a collaborative, transdisciplinary approach to mainstream biodiversity into LTIs. IENE's network includes a diverse range of stakeholders—decision-makers, road planners, environmental authorities, and researchers. IENE provides an independent, international and interdisciplinary platform for developing and exchanging expert knowledge, information and latest advancements, fostering the development of a safe, integrated and ecologically sustainable pan-European transport infrastructure. One significant way IENE supports the integration of biodiversity into transport networks is through the publication of expert knowledge, showcasing the latest solutions and best practices. A relevant example is the "Biodiversity & Infrastructure: A Handbook for Action" (Rosell et al. 2023), produced under the Horizon 2020 BISON project (https://bison-transport.eu/). This handbook synthesizes over 20 years of experience addressing the impacts of transport infrastructure on biodiversity, offering practical guidelines for designing, upgrading, and operating ecologically sensitive infrastructure. The Handbook is divided into seven chapters covering policy and planning, the mitigation hierarchy, landscape integration, infrastructure solutions, assessment and monitoring, and maintenance. It encourages collaboration across disciplines, having been developed by 50 authors and reviewed by 30 experts from both ecological and infrastructure sectors. By using a shared terminology, it bridges the communication gap between ecologists and infrastructure professionals, fostering mutual understanding and cooperation. It refers to both European and global guidelines to ensure its continued relevance in addressing the challenges of biodiversity protection and climate adaptation. Designed to be flexible and adaptable, the Handbook can be updated as new challenges and solutions arise.

IENE members are actively engaged in developing cutting-edge solutions to prevent wildlife-vehicle collisions, such as alert and warning systems for both drivers and animals (Huijser et al. 2015; Seiler and Olsson 2017), and tools for monitoring the presence of wildlife on roads and railways (Carvalho et al. 2017; Bhardwaj et al. 2020; Shilling et al. 2020). The exchange of such innovative solutions, along with best practices and knowledge aimed at making LTI more sustainable and ecologically friendly, is facilitated through various IENE events, including its biennial conferences held across Europe (Sjölund et al. 2022). At the IENE Conference in September 2022, hosted in Cluj-Napoca, Romania, numerous ground-breaking studies were shared, with a selection of emblematic research published in this Special Issue titled "Connecting People, Connecting Landscapes." These studies tackle crucial topics related to the integration of biodiversity into transport networks, such as the effectiveness of wildlife crossings, the preservation of ecological connectivity at the landscape level, environmental impact assessments for both old and new LTI, animal-vehicle collision prevention, wildlife exclusion from transport networks, and the role of LTI in providing habitat and refuge for wildlife.

IENE has systematically addressed these topics over time, providing guidance for both infrastructure planners and decision-makers to integrate ecological considerations into their projects and policies. For instance, the Global Strategy for Ecologically Sustainable Transport and Other Linear Infrastructure (Georgiadis et al. 2020) outlines specific objectives and principles for governments and organizations to mainstream biodiversity and ecological connectivity in transport infrastructure development. This strategy emphasizes the need for multi-sectoral collaboration, proactive policies, appropriate legal frameworks, and the application of innovative, scientifically sound solutions. The outcomes of the 2022 conference, including the Cluj-Napoca Conference Declaration, further reinforced this direction, calling for strengthened cooperation among stakeholders to ensure that transport infrastructure development aligns with biodiversity conservation and landscape connectivity goals.

Declaration of the IENE 2022 International Conference
Cluj–Napoca, Romania, 19–23 September 2022

We, the participants to the IENE 2022 International Conference in Cluj–Napoca, Romania, acknowledge that:

1. While the transport sector (including infrastructure and energy networks) is crucial to the development of human society, a diverse and functional natural environment is the prerequisite not only for our well-being but ultimately for our survival as a species.
2. The centuries of intensive transport development in Western Europe also taught us how detrimental for nature, and especially wildlife and the coherence of ecosystems and landscapes, this infrastructure could become if it is designed, built or operated in an unsustainable way.
3. The transport sector is closely connected, directly or indirectly, to the five main direct drivers of biodiversity loss[1]:
   i. **Land- and sea-use change**, by irreversibly fragmenting habitats and populations and by increasing wildlife mortality risks and sealed soil, changing the structure and functionality of ecosystems and generating a cascade of changes at landscape levels;
   ii. **Direct exploitation of organisms**, by facilitating access to previously remote natural areas/roadless areas and overexploitation of animals, plants and other organisms, mainly via gathering, logging, hunting and fishing;
   iii. **Climate change**, by increasing green-house gas emissions during the construction and use of the infrastructure;
   iv. **Pollution from all sources**, emitted during the construction and use of the infrastructure; and
   v. **Invasive alien species**, facilitating their introduction and spread.

1 Brondizio ES, Settele J, Díaz S, Ngo HT (Eds) (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. IPBES secretariat, Bonn, Germany. https://doi.org/10.5281/zenodo.3831673

As these drivers are reinforced by underlying causes such as the lack of consistency across sectoral policies, subsidies, and between regulations, there is a strong and urgent need for integrated sustainable approaches and an adequate and effective governance.

4. Although during the past decades the concern about the impacts of the transport sector led to better approaches, adapted techniques and increased expertise on how to plan, build and operate sustainable transport networks, the past issues remained, making Western Europe one of the most fragmented landscapes worldwide, which has led to several countries responding by implementing vast and costly defragmentation programs and plans.
5. In contrast with Western Europe, the Eastern part of the continent is rightfully demanding urgent extension and modernization of its transport infrastructure. At the same time, this area of Europe still holds unique natural and cultural values, productive landscapes and functional ecosystems as a result of predominantly extensive use of natural resources.
6. In the current political and socio-economic context, Eastern Europe and regions like the Balkans and the Black Sea are being presented with a unique possibility: to develop transport infrastructure that does not cause a devastating and costly fragmentation of nature, making the best use of existing knowledge accumulated over the last decades.
7. Moreover, Eastern Europe has the opportunity to become a reference region for overall sustainable development, especially in the critical context of climate change, water shortage, land degradation and biodiversity loss.

WE CALL FOR URGENT ACTIONS, FROM POLICY TO PRACTICE, and invite the entities at all levels (local, national, European and international) of governments, conventions, organisations, academia, institutions, businesses, transport planners, constructors and operators, networks, experts, funders, mass-media and civil society to foster cooperation, in order to:

1. Adopt sustainability in transportation development across the spectrum of human activities in the 21st century as essential under four basic pillars:
   i. The well-being of societies;
   ii. The resilience of healthy economies;
   iii. Environmental quality and safety and the link with effective biodiversity conservation;
   iv. Keeping the impacts of human activities on the environment reversible.
2. Recognize that safeguarding ecological connectivity is a key aim and a major challenge for the transport sector, which needs to be addressed in spatial planning in collaboration with other sectors (i.e., other infrastructure, agriculture, forestry, tourism, hunting, water management, protected areas, etc.).
3. Include as a key objective for sustainability the avoidance of fragmentation of nature and landscapes in all developing activities, in accordance with relevant strategic policy documents and technical recommendations[2].

2 i. The Convention on Biological Diversity 2018 decision on mainstreaming biodiversity in developing sectors including transport.

4. Adopt and implement the principles of the **IENE Global Strategy for Ecologically Sustainable Transport and other Linear Infrastructure**, namely:
    i. Create **a strong policy and legal framework** on safeguarding landscape connectivity as a primary concern for any project scale including regulatory requirements through standardization of tools, methods, etc.;
    ii. Begin with **strategic planning** with the implementation of "Avoidance – Mitigation – Compensation" mitigation hierarchy[3];
    iii. Follow an **ecosystem approach** based on the "Precautionary Principle"[4] respecting the value of natural capital and ecosystem functions and services;
    iv. Evaluate that **any case is a unique case.** Each project is site-and species-specific and therefore unique. Mitigation should be based on scientific and best available local knowledge without "copy paste" from other projects and cumulative impacts of other local projects should be taken in to account;
    v. Enhance **multi-disciplinary and cross-sectoral cooperation**;
    vi. Implement **the responsible polluter pays principle** not only from the pollution perspective, but also taking into consideration the impacts on biodiversity and ecological connectivity as well as ethical and transparency concerns;
    vii. Include **long life effective maintenance** and sufficient monitoring in all planning and budgeting of transport and other developing projects.
    viii. Create **climate change resilient infrastructure**;
    ix. Plan and manage **adaptable infrastructure habitats** to fulfill their potential as positive biodiversity refuges and ecological corridors;
    x. Establish **environmental supervision and monitoring** of the effectiveness of transport infrastructure features on wildlife permeability in all phases of programs, plans and projects;
    xi. Promote a **culture of learning** to develop continuous evaluation and exchange of knowledge and experience.

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ii. The Sustainable Development Goals include addressing biodiversity loss and securing ecological connectivity as essential drivers for sustainability.

iii. The United Nations plans for an active Restoration Decade through 2030.

iv. The EU Strategies for Biodiversity and Green Infrastructure.

v. The EU Green Deal and the implementation of Europe-wide Restoration Goals.

vi. The EC technical guidance on the application of "do no significant harm" under the Recovery and Resilience Facility Regulation, C(2021) 1054 final.

vii. The Carpathian Convention's initiatives on Sustainable Transportation, Biodiversity conservation and ecological connectivity.

viii. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) global assessment (2019) found that of the five pressures on biodiversity, the most important on terrestrial environments was land-use change, which may include deforestation, intensification in agricultural management, or habitat fragmentation.

ix. Experts from the Intergovernmental Panel on Climate Change (IPCC) and IPBES met in 2020 and concluded that none of the issues identified by these two platforms can be solved if they are not addressed together.

3 See the Glossary of the Working Group on No Net Loss of Ecosystems and their Services: https://ec.europa.eu/environment/nature/biodiversity/nnl/index_en.htm

4 https://eur-lex.europa.eu/EN/legal-content/summary/the-precautionary-principle.html

5. Develop an urgent common framework of priority actions from policy to practical implementation of evidence-based solutions to mainstream biodiversity into a sustainable transport sector, such as:
   i. Support the appropriate political will for taking decisions based on criteria of the four pillars of sustainability and biodiversity conservation needs;
   ii. Think globally and implement policies locally while filling the gaps and overcoming barriers that have been highlighted by relevant transport & ecology projects (e.g., BISON[5], TRANSGREEN[6], ConnectGREEN[7], SaveGREEN[8], HARMON[9], among others);
   iii. Cooperate to enable the coexistence of ecological and transport corridors through the implementation of EU TEN-G[10], TEN-N[11] and TEN-T[12] Strategies while effectively sharing experience and know-how between countries and entities across Europe and globally;
   iv. Develop cross-sectoral tools and management practices for effectively protecting the coherence of the ecological networks (e.g., NATURA 2000, Emerald) and the integrity of their component sites and of other protected areas (e.g., parks or reserves);
   v. Proactively produce and use the scientific and practical knowledge to promote innovative and sound evidence-based sustainable solutions and make use of updated data bases, modern standards and innovative methodologies;
   vi. Include in the necessary assessments (e.g., Strategic Environmental Assessments, Environmental Impact Assessments, Appropriate Assessments, Climate Change, or Water Framework Directive Assessments) independent scientific expertise and environmental supervision while involving the local society and the relevant stakeholders;
   vii. Imp lement the appropriate measures to avoid, reduce and compensate the impacts on biodiversity, based on multidisciplinary cooperation between social scientists, environmentalists and engineers in order to achieve infrastructure sustainability, resilience and acceptability at landscape level.

Cluj-Napoca, Romania, September 2022

5 https://minuartia.com/en/the-horizon-2020-project-bison-presents-final-results-at-the-european-council-in-strasbourg/

6 https://dtp.interreg-danube.eu/approved-projects/transgreen

7 https://dtp.interreg-danube.eu/approved-projects/connectgreen

8 https://dtp.interreg-danube.eu/approved-projects/savegreen

9 https://green-web.eu/projects/harmon/

10 https://environment.ec.europa.eu/topics/nature-and-biodiversity/green-infrastructure_en#:~:text=Green%20infrastructure%20has%20been%20defined,example%2C%20water%20purification%2C%20improving%20air

11 https://www.eea.europa.eu/publications/building-a-coherent-trans-european

12 https://transport.ec.europa.eu/transport-themes/infrastructure-and-investment/trans-european-transport-network-ten-t_en

Additional information

Conflict of interest

The authors have declared that no competing interests exist.

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Data availability

All of the data that support the findings of this study are available in the main text.

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Research Article

Use of linear transportation infrastructure rights-of-way as an ecological shelter: National asset estimate and stakeholder involvement

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Abstract

For a number of years, the rights-of-way (ROW) of several types of linear transportation infrastructure (LTI), such as roads, railways, waterways and power lines, have been regarded as possible shelter for biodiversity, notably local flora and entomofauna. For developing an informed general management policy of such an opportunity for species conservation and landscape connectivity, a fundamental prerequisite is to be aware of the ecological shelter potential available within LTI rights-of-way on a national scale. By considering the primary operating constraints of LTIs and their geometric characteristics, a GIS-based method was designed to approximate the linear extent and surface area of potential ecological shelter (PES), i.e. where actions could be implemented to provide sustainable shelter focused on local flora and entomofauna. At the scale of Metropolitan France, the minimum total surface area of PES amounts to 2,026 km^2; and its network spans over 88,094 km (39% power lines, 34% railways, 18% roads and 9% waterways). The State is the primary landowner of PES along operated networks (particularly railways) however over half of the assets (53.8%) fall under the responsibility of local authorities, concessionary companies and private landowners (notably power lines). These findings highlight the necessary involvement of State together with LTI operators, local stakeholders and landowners through partnerships aiming to implement the ecological shelter function of rights-of-way.

Key words: Biodiversity, conservation, geographic information system, habitat, management, network, Potential Ecological Shelter

Introduction

Rights-of-way of LTIs and safeguarding biodiversity

The term right-of-way (ROW) defines the entire width of the reserved strip of land on which linear transportation infrastructure (LTI) is built (van der Ree et al. 2016). For a number of years, the rights-of-way for several types of LTI, such as roads, railways, waterways and power lines, have been regarded as possible shelters for biodiversity, in particular across those landscapes impacted by human activities which have led - or are still leading - to the destruction of natural habitats and corridors (Baudry et al. 1995; Gardiner et al. 2018). To some extent, rights-of-way would be expected to improve the functioning of such damaged landscapes, by means of restoring connections between natural networks (hedges, riverside vegetation) or isolated habitats (Meunier et al. 1998; Michel et al. 2015).

Similar to natural corridors (e.g. rivers), manmade corridors like LTI rights-of-way can exhibit five main functions for species: habitat, conduit, source, barrier and sink (Burel and Baudry 1999; Forman et al. 2003). The role played by a corridor, whether or not it is natural, depends on the biological characteristics of the considered species, and also on its structure and place in the landscape (Burel and Baudry 1999).

The benefit of rights-of-way as a potential habitat is particularly relevant for native flora and entomofauna. Besides, there are numerous considerations numerous considerations at play, particularly concerning the conservation of threatened species, connectivity within landscapes (along rights-of-way and with neighboring green networks) and the provision of ecosystem services (De Redon de Colombier 2008; Hopwood 2008; Wagner et al. 2014; O'Sullivan et al. 2017; Villemey et al. 2018; Guo et al. 2022). Furthermore, in the context of climate change, biodiversity preserved and smartly managed within rights-of-way can provide opportunities for nature-based solutions (NbS) aimed at reducing the risks and consequences of extreme events (high/low temperatures, high precipitations, droughts, high winds…) on transport infrastructures, their quality of service and their users (Blackwood et al. 2022).

It can therefore be expected that implementing suitable management inside rights-of-way in the aim of developing their potential functions as habitats, conduits and possibly sources for re-colonizing neighboring degraded landscapes (Burel and Baudry 1999), could yield positive impacts. Such impacts would be felt not only at the local scale but also to a broader extent by limiting and compensating for the general decline of so-called common species, such as weeds and flying insects as well as downstream beneficiaries like birds (Hallmann et al. 2017; Richner et al. 2017; Stanton et al. 2018).

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services points out the immediate opportunities to improve the current conditions of pollinators and maintaining the pollination service in road verges (IPBES 2016). This could be the case for all kinds of LTI rights-of-way and might also benefit other entomofauna species (Cardoso et al. 2020). With the ability to undertake general or specific actions within rights-of-way, LTI operators can become direct actors in local flora and entomofauna conservation and in the restoration of connectivity within landscapes. As an example and more broadly, in Sweden, the concept of 'responsibility species' was proposed by Helldin et al. (2015) in the aim of actively involving road and railway operators in the conservation of red-list species thanks to infrastructure verges.

On a global scale, states have committed to following the Kunming-Montreal Global Biodiversity Framework (UN Environment Programme, 2022), which is a contribution to the achievement of the 2030 Agenda for Sustainable Development (UN General Assembly, 2015). The framework calls for respect for the integrity of all ecosystems. Leveraging rights-of-way of LTIs to enhance biodiversity, ecosystem functions and services, ecological integrity and connectivity, for the benefit of people and nature in degraded ecosystems, meets three targets (n°2, 11 and 12) for action by 2030. Hence such an initiative can support any State to comply with the Global Biodiversity Framework agenda.

Estimate of the national asset of potential ecological shelter

In order to develop relevant and consistent actions within ROW, and beyond, a general management policy in favor of biodiversity through the rights-of-way of various LTIs, it is essential to know their extent and the surface areas that can be realistically and effectively harnessed for the species of the local flora and fauna. This can be achieved by considering their structure and surroundings. However, restoring and/or developing corridor functionalities thanks to ROW cannot be an objective for all territories crossed by LTI networks. This point is notably true in the case of urban and industrial areas where, in addition to several local disturbances, rights-of-way are generally narrow, highly fragmented or perhaps nonexistent. Hence, extrapolation from the total LTI length (provided by annual compendia for example (EC 2022; CGDD 2019) cannot be the way to approximate the potential of ROW actually available at the scale of a large territory (e.g. country, region) and cannot therefore provide a reliable support to develop implementation strategies there.

Some LTI operators publish figures of the total surface area of their rights-of-way (Michel et al. 2015). Estimates however rely on their in-house approaches, which vary from one operator to the next and in most cases are not fully explicit. Some estimates comprise all green spaces, including those linked to human activities (e.g. motorway rest and service areas), while others do not. Some operators apply a fixed or variable ROW width around their LTI, while others use compilations from land registry maps. Lastly, not all operators have this estimate on ROW surfaces and moreover are unable to measure them over the short term. As a result, it is not possible to derive a complete, homogeneous and clear picture of the actual situation at the national, regional or local scale.

An analysis of cartographic databases available today (knowledge of LTI networks and crossed environments), combined with a consideration of the geometric criteria of LTI cross-sections and certain maintenance rules for the various types of ROW, offers one way to acquire this knowledge. This approach can provide a transparent estimation method, using well-founded uniform criteria throughout the assessed territory. It is adaptable to the main variations in LTI characteristics from one territory to another and to particular sections of networks. Hence, it can be reproduced at all territorial scales, regardless of their diversity, and handled by LTI operators and land managers from any territory equipped with mapping tools.

Consistent with this approach and its aims, a GIS-based method was designed to approximate, on a national scale, the length and surface area of rights-of-way parts from various LTIs which could be dedicated to local flora and entomofauna: a space hereafter referred to as potential ecological shelter (PES). PES concerns a part of biodiversity whose presence in ROW is not detrimental to the normal and safe exploitation of infrastructures. Additionally, areas considered as PES exclude ROW parts whose maintenance requirements are not compatible with the respect of species' ecological needs and specimens' integrity (e.g. road and railway proximity strips), ROW parts that could create isolation or traps (e.g. road medians and interchanges), and ROW sections located in environments that are not conducive to the establishment of landscape connectivity with the surroundings (urban and industrial contexts). The development work was carried out and assessed in the French context, where cartographic databases dedicated to LTI networks and land uses are available, where the general view of ROW surface potential was heterogeneous and fuzzy, and where some figures from operators could serve as reference values to assess the GIS estimation process itself.

For many LTIs, the outer ROW boundaries often have complex shapes (e.g. property lines, topographical discontinuities, flood zones associated with watercourses), which cannot be properly and easily handled when estimating at the national scale. The ambition of the estimate was thus not to target the total maximum surface of PES, as acuity would be too low to demonstrate how the approach clarifies and supports decision-making. A more realistic perspective, which can offer greater direct utility for many situations in today's national and regional contexts, aims to assess the value of the minimum total surface of PES available at national scale. This means a guaranteed and more accurate floor value, as it has been calculated from the minimum incompressible width of different types of ROW. This constitutes a reliable minimum basis for developing policy and actions, while maintaining the possibility of refining data at lower levels with local stakeholder participation.

The national asset estimation method is presented below, along with the principles and tools. The results for all types of LTIs are detailed and discussed, notably in light of minimal pre-existing figures and stakeholders who could enable ROW become biodiversity shelters. This emphasizes the shared involvement among LTI operators as well as local stakeholders and landowners to implement the ecological shelter function of rights-of-way.

Materials and methods

Cartographic tools

The geographic information processing software used for this study was QGIS 3.4. The database employed for LTI networks was BD TOPO® (provider National Geographic Institute, IGN; design scale: 1:25,000) (IGN 2019). For the road, railway and power line network maps, the data were extracted from "Road network", "Rail network" and "Energy transport network" subfiles (version 2.2, January 2019), and for waterway network map, from the "Hydrography" subfile (version 3.0, April 2019) which allowed checking the actual navigability of each waterway segment. The national waterway operator (VNF) database served to delineate the State-owned domain. Figures from recent compendia on the total length of transport networks (Michel et al. 2015; CGDD 2019; EC 2022) were used to calibrate the compilation method with databases and ensure reliability for subsequent calculations (avoidance of double counting and omission of entities).

The database used for the forest cover was BD FORET® version 3 (provider: IGN; design scale: 1:25,000; update: November 2018 (IGN 2018)). This resource records the various types of plant and forest formations, according to the majority of species observed over areas larger than 0.5 hectares and at least 20 m wide, primarily based on the rate of plant cover (e.g. ≥ 40% = closed forest). The Corine Land Cover database, hereafter referred to as CLC (provider: European Environment Agency; design scale: 1:100,000; update: 2012 (CGDD 2016)), was used for delineating the urban and industrial spaces.

Delineation of urban and industrial areas

Objects entitled “Continuous urban fabric”, “Discontinuous urban fabric”, “Industrial or commercial units and public facilities” and “Port areas” in CLC were collated and around the resulting urban polygons, a 200-m wide buffer was applied. In CLC, objects such as “Road and rail networks and associated areas” and “Rivers and waterways” often pass through the urban fabric without being considered part of it, hence becoming merged with it. The 200-m wide buffer was intended to cover (i.e. mask) the presence of LTI networks within the urban/industrial fabric, hence to exclude them from the ROW surface calculation. Its size was set as the minimum width covering the various types of LTI rights-of-way in a large city: tests were carried out on the conurbations of Paris (48.856°N, 2.352°E), Bordeaux (44.837°N, -0.579°W; 780,000 inhabitants) and Nantes (47.218°N, -1.553°W; 650,000 inhabitants).

LTIs and their specific ROW width

For each type of LTI, Fig. 1 schematically summarizes the location and extent of ROW parts considered in the estimate.

Power lines

Two types of transmission lines were considered for the calculation: high-voltage power lines (63 to 90 kV – hereafter referred to as HV lines) and very high voltage lines (225 to 400 kV – referred to as VHV lines). The ROW segments referred to as “Unspecified”, “Unknown” and “Power down” in the database, as well as a short 150-kV segment, were not considered as they account for just 5 km at the national scale.

For power lines, a continuous linear land right-of-way only appears when crossing wooded areas (Fig. 2). In open contexts, their ground footprint is limited to the pylon footprint. Thus, for the PES estimate, the power line network is considered where it passes through forests and the width potentially available as an ecological shelter is the full ROW width.

In light of technical and safety requirements, the typical right-of-way width for a HV line where it cuts through woodland is 30 m, while VHV lines are assigned a 50-m width (Michel et al. 2015; François et al. 2018). These values have been verified on aerial views from the public databases Géoportail (www.geoportail.gouv.fr) and Google Earth (www.google.fr/earth/) in different forested areas in France: Ardennes (Northeast), Limousin (Center-West) and Landes (Southwest).

Waterways

Two types of waterways were considered for this calculation: shipping canals (artificial waterways), and navigable rivers (natural waterways). For both, the ROW can extend far from the bank, depending on local geomorphological conditions.

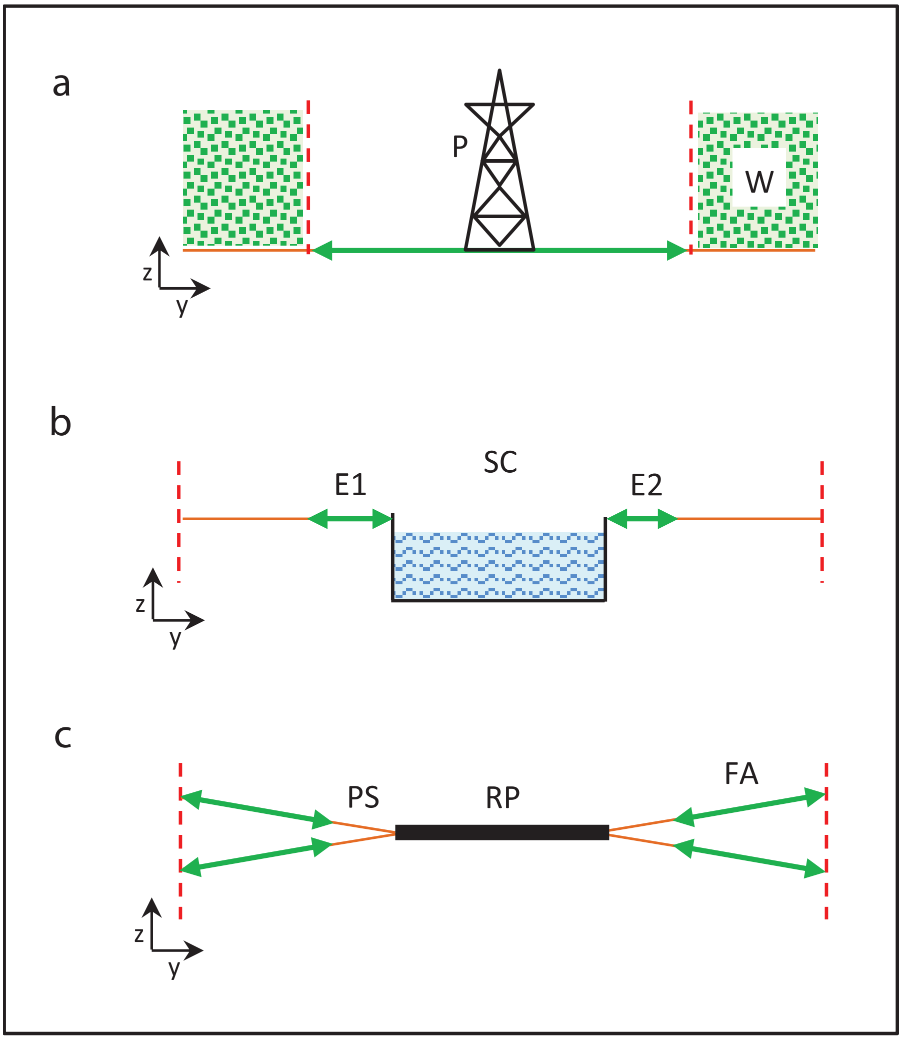

Figure 1. ROW cross-sections of **a** power lines **b** waterways (case of shipping canal) **c** railways and roads. Caption: Dotted red line: side boundaries of the entire ROW; Brown line: ground surface; Double green arrows: extent of ROW portions considered in the estimate; P: pylon; W: wooded area; SC: shipping canal; E1 and E2: easements of waterways; RP: railway or road platform; PS: proximity/safety strip; FA: far area.

Figure 2. Dry grassland hosting a protected species (*Gagea pratensis*) in a power line ROW, in the center of France (47.8644°N, 1.8076°E).

The rules for delineating the outer boundaries are defined by law (Etrillard 2019). At this study scale, it was not possible to consider a detailed analysis of the actual width of all waterway rights-of-way, nor was it realistic to estimate a representative average width, for both artificial and natural waterways.

In contrast, for technical servicing purposes, two regular strips of land are reserved on both sides of waterways (Fig. 3). The broader one is the towpath easement (7.80 m wide – i.e. 24 feet), while on the opposite bank a 3.25 m-wide land strip (10 feet) is reserved for operators, fishermen and pedestrians. With the aim of providing a realistic floor value, the minimum legal ROW width (rounded to 11 m) was thus adopted for this calculation.

Figure 3. A shipping canal right-of-way with its towpath in the foreground, in the north of France (50.9400°N, 2.2622°E).

Railways

Railway types considered in the estimate

Four types of railway infrastructure were considered for this estimate: high-speed rail network, main railway lines, service track, and unused track.

In BD TOPO®, high-speed lines are defined as being reserved for high-speed trains and are referred to as LGV (for *Ligne à Grande Vitesse*). The main railway lines correspond to all lines in operation that provide regular or seasonal service transporting passengers or goods, with the exception of LGVs. The database solely considers service track to extend more than 200 m in length and the definition excludes track segments included in a bundle of lines more than 25 m wide (e.g. marshalling yards), as well as subterranean track. As regards unused track (defined as declassified and closed to any kind of traffic), the database also solely considers segments more than 200 m long; subterranean segments are also excluded.

All railway lines are bordered by so-called “proximity strips”, where for technical and safety reasons vegetation is strictly controlled throughout the year to keep it short, by means of mowing or chemical weed treatment (Michel et al. 2015). For LGVs, this strip is 4 m wide on both sides, whereas 3 m is the width for both the main railway lines and service track.

In order to more easily maintain higher speeds, the longitudinal profile of LGVs needs to be as flat as possible, hence the land’s natural relief may be

Figure 4. A LGV right-of-way with embankment in a hilly landscape, in the east of France (47.3370°N, 6.0075°E).

highly modified by earthworks. Compared to the main railway lines, this leads to higher embankments and deeper cuts to enable high-speed lines to pass through the landscape (Fig. 4). Hence, all along LGVs, the ROW width varies to follow changes in the natural relief. Maximum width is found at both the top of cuts and bottom of embankments; the minimum occurs where the longitudinal profile of the track (called the red line) coincides with the natural topography.

Estimates of railway rights-of-way average widths – Influence of topography

In France, the red line very rarely coincides with the natural relief. For the purposes of this estimate, an average reference width for the LGV right-of-way in France was sought. In order to estimate a realistic value, previous studies on geometry and earthwork slopes (Ginot et al. 2010; Fargier 2013) were combined with aerial view observations (Géoportail cross-referenced with Google Earth), for slightly rolling landscapes crossed by LGVs. It appears that the minimum side width for LGVs can be considered equal to 17 m (value for flat zones). For slightly rolling areas, the average LGV side width amounts to 41 m (derived from: vertical variations = ±6 m; earthwork slopes = 2/1; plus the previous 17 m). Main railway lines and service track (as well as unused track) do not necessitate such major earthworks. Based on the same technical references and map observations (Géoportail and Google Earth), their average side width was set equal to 15 m.

Consequently, for LGVs, depending on topography, the average ROW width potentially available as an ecological shelter (i.e. the so-called "far area"), is 9 m (minimum value corresponding to a theoretically flat topography) and 33 m (average relief). For main railways, service and unused track, this value is 9 m (average relief).

Roads

Road types considered in the estimate

Three types of road infrastructure were considered for this assessment: the motorway network, the so-called "quasi-motorway" network, and the dual carriageway road network.

Figure 5. A motorway right-of-way (proximity strip and far area), in the west of France (47.0602°N, 1.4243°W).

In BD TOPO®, motorways are defined as "roads without crossings, accessible only at points set up for this purpose and reserved for power-driven vehicles"; and are classified as such by decree of the Council of State. Quasi-motorways satisfy the same technical definition as motorways (carriageways separated by a central median, no at-grade intersection with the rest of the road network) but are not officially classified as such. Dual carriageway roads have two pavements separated by a physical obstacle that may open at grade intersections. The presence of intersections prohibits them from being classified in either of the previous categories.

Roads are bordered with a strip of land where, for technical and safety reasons, vegetation is strictly controlled throughout the year (Michel et al. 2015). The width of this safety strip (Fig. 5) can vary slightly from one manager to the next, but for the main road networks (motorways, quasi-motorways, dual carriageway roads), the minimum is 4 meters (François and Le Féon 2020).

Similar to LGVs, to allow for higher speeds, motorway longitudinal profiles also result in many embankments and cuts in the natural topography. However, motorways can comply with steeper slopes than LGVs (maximum slope of 6% vs. 3.5% (Ginot et al. 2010)), which leads to smaller cuts and embankments, hence narrower ROW in rolling landscapes.

Estimates of road right-of-way average widths – Influence of topography

An average reference width for motorways and quasi-motorways was sought. To estimate a realistic average width for their rights-of-way in France, documentation on their geometry and earthwork slopes (Ginot et al. 2010) was combined with aerial view observations (Géoportail and Google Earth) for slightly rolling landscapes. For (quasi-)motorways, the minimum value of total ROW width thus amounts to 8 m (with no topographic changes) and for medium hilly areas, the average width is 24 m (derived from: vertical variations = ±4 m; earthwork slopes = 2/1; plus the previous 8 m).

In order to characterize the average total ROW width for dual carriageway roads, map observations were carried out over 44 cross-sections from 7 such roads across the country. This led to an average minimum right-of-way width of 10 m. In considering the ROW width variations along these road sections (often above 20 m and up to 25 m) and the average width for each one (9 to 19 m), a value of 16 meters has been selected to characterize the average total right-of-way width for dual carriageway roads.

It appears that for (quasi-)motorways, the average ROW width potentially available as an ecological shelter (the "far area") equals zero in the theoretical case of flat topography, and 16 m in the case of average relief. For dual carriageway roads, figures are respectively 2 m and 8 m. These areas are part of road verges, defined as the vegetated area adjacent to roads (van der Ree et al. 2016).

Road medians and interchanges

The re-vegetation of road medians with attractive plants in the aim of creating a habitat and/or kind of stepping stone between both sides of the infrastructure is regarded as a counterproductive initiative with respect to flying insects, such as bees, butterflies and dragonflies, responsible for more collisions (Keilsohn et al. 2018; François and Le Féon 2020). Accordingly, this kind of ROW has not been included in the calculation of PES. Moreover, in France, medians are generally far narrower than twice or even one times the safety strip (i.e. 4 m). With respect to this same crossing issue, motorway interchanges and their associated verges were not considered in the estimate: for one thing, they create isolated green areas and for another, their inclusion in the calculation would falsely increase the length of road sections with PES. All these sections were differentiated thanks to their referencing as "Ramp" in BD TOPO®.

Results

The detailed figures for the entire estimate are provided in Table 1. For each subtype of transport infrastructure are given: the length of potential ecological shelter, the share of the total PES network this represents, the width of ROW considered in the calculation and the surface area of ecological shelter.

Table 1. Features of the potential ecological shelter offered by the various LTI networks in Metropolitan France.

LTI type	Subtype	Length (km)	Share of total linear (%)	Width (m)	ROW share[a]	Surface area (km^2)
Power lines	High voltage	13,347	15.2	24	Entire width	320
	Very high voltage	20,652	23.4	50	Entire width	1,033
Waterways	Shipping canals	3,603	4.1	13	E1 + E2	47
	Navigable rivers	3,991	4.5	13	E1 + E2	52
Railways	High-speed lines	1,839	2.1	33	FA	61
	Main railways	24,035	27.3	9	FA	216
	Service track	1,430	1.6	9	FA	13
	Unused track	2,868	3.3	15	PS + FA	43
Roads	Motorways	10,276	11.7	16	FA	164
	Quasi-motorways	3,481	4.0	16	FA	56
	Dual carriageway roads	2,572	2.9	8[b]	FA	21

[a] See Figure 1.
[b] The dual carriageway roads for width measurements were N4 around Vitry-le-François (48.726°N, 4.585°E), N12 around Dreux (48.736°N, 1.370°E), N42 around Boulogne-sur-Mer (50.725°N, 1.613°E), N154 around Evreux (49.027°N, 1.511°E), CD83 around Colmar (48.079°N, 7.358°E), CD 775 around Segré (47.689°N, 0.868°W), and CD 824 around Dax (43.708°N, 1.051°W).

Linear extent of potential ecological shelter

Linear extent of PES along LTI networks

Power lines provide the greatest length of PES (33,999 km in all). This value accounts for approximately one-third of the total length of transportation networks managed by the national operator (100,000 km in Michel et al. 2015; 105,500 km calculated from BD TOPO®). The total linear extent of PES along waterway network (7,594 km) is nearly equally split between shipping canals (47%) and navigable rivers (53%). Railways provide the second greatest length of PES (30,172 km total). LGVs constitute just 6% of the total for railways, while the vast majority (79%) stems from the densely-branched main railway network. Unused track is next with nearly 10% of the total, and service track offers the least length, with some 5%. Regarding roads, the total length of PES (16,329 km) is principally provided by motorways (63%). Aside from the classification issue (motorway vs. quasi-motorway), the overall length for this type of infrastructure (i.e. (quasi-)motorway) amounts to 13,757 km, i.e. 84% of the total potential ecological shelter along road network. Dual carriageway roads cover the remaining 16%. As a result, the total length of PES along the four LTI networks in Metropolitan France is 88,094 km.

Share of the different LTIs in the total PES linear extent

Power lines are the most extensive part of the total PES network (38%, Fig. 6), slightly ahead of railways (34%), with the main railway lines alone accounting for 27% (the leading LTI subtype). By definition, the power lines of interest for PES are generally localized in remote large forested areas, whereas the densely-branched main railway network encompasses almost all parts of the national territory. The road network is the next most extensive LTI (at 19% of the total), of which (quasi-)motorways represent by far the majority share (a combined 16%). Lastly, the waterway network accounts for 9% of the total yet is more concentrated in certain regions (e.g. Northeast). This ranking with respect to network extent differs from that based on the number of studies and research

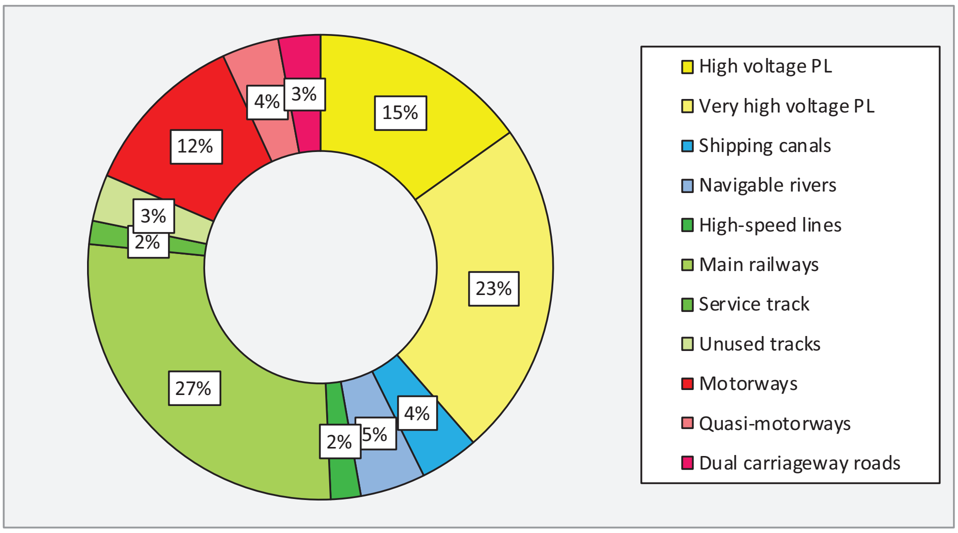

Figure 6. Share of the different LTIs and their subtypes in the total linear extent of PES at national scale.

carried out until now on LTI rights-of-way for insects in temperate regions, where roads come in first place, followed by power lines, with both railways and waterways receiving limited attention (Villemey et al. 2018).

Surface area of potential ecological shelter

Surface area of PES along LTI networks

The surface area of ROW under HV lines in forest corridors amounts to 320 km^2. The value for VHV lines (longer network and wider cuts) is significantly larger: 1,033 km^2. The total surface area for power line ROW represents 36% of the total ROW surface reported by the national operator (i.e. 4,000 km^2 in Michel et al. 2015). The surface areas of potential ecological shelters bound to shipping canals and navigable river easements accounts for 47 km^2 and 52 km^2, respectively. Considering the average cross-section of LGVs through slightly rolling landscapes, the surface area of PES amounts to 61 km^2. Due to the large extent of the main railway network across interurban areas, the surface area of PES increases to 216 km^2. With the same typical cross-section (i.e. 9 m of ROW width), service tracks account only for 13 km^2. Unused tracks (15 m of ROW width) account for 43 km^2. Considering the average cross-section of motorways and quasi-motorways through slightly rolling landscapes, the surface area of PES amounts to 164 km^2 and 56 km^2, respectively. Lastly, for dual carriageway roads, considering the average cross-section, the surface of PES equals 21 km^2. As a result, the total area of PES along the four LTI networks in Metropolitan France is 2,026 km^2.

Share of the different TLIs in the total PES surface area

Power lines provide the highest share of the total PES surface area (67%, Fig. 7) with VHV power lines as leading LTI subtype (51%), far ahead of railways (17%), with the main railway lines alone accounting for 11%. The road network is next (12%) of which (quasi-)motorways represent by far the majority share (a com-

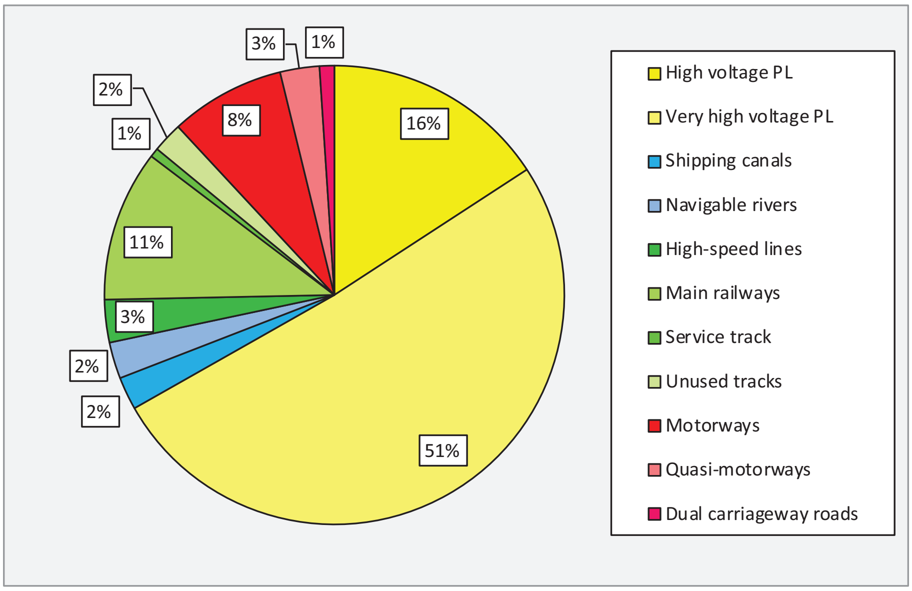

Figure 7. Share of the different LTIs and their subtypes in the total surface area of PES at national scale.

bined 11%). Lastly, the waterway network (for which only easements have been considered) accounts for 4% of the total surface area.

This PES surface floor value is just a small fraction (i.e. about a third) of the surfaces indicated by the various LTI operators for their full ROW asset, which amounts to approximately 5,900 km^2; excluding the surface area of the national road network asset, not yet accurately estimated (NNCM 2009; Michel et al. 2015). These large figures include narrow road verges, proximity strips, ROW located in urban and industrial zones, isolated surfaces inside interchanges and road medians.

Discussion

Potential ecological shelter bound to power line network

Linear extent and surface area features

The larger proportion of VHV lines (20,652 km), compared to HV lines (13,347 km), stems from the former being located further from power distribution (i.e. urban) areas, thus more conducive to crossing large wooded areas. Besides, just 25% of the total length of HV lines (53,500 km calculated from BD TOPO®) are located in forests, compared to 40% for VHV lines (total length: 52,000 km). The electricity transmission operator (RTE), a single national entity, does not own the ROW, which remains the property of the landowner (private, municipal or national forests) (Etrillard 2020).

Current management of ROW

As a general rule, the operator substitutes for the landowner in maintaining ROW vegetation. He determines the maintenance techniques and tree cuts performed to ensure the safety of the power lines, yet the cut vegetation remains the landowner's property (Etrillard 2020). Rotary-slashing is the most widespread maintenance practice today for ROW vegetation, with one pass every 3–5 years, which often benefits fern colonization to the detriment of plant diversity (François et al. 2018). Other ROW maintenance modes are possible (oriented for instance towards pastureland, hay meadows, conservatory orchards or the restoration of natural open habitat) and generally based on local partnerships (Godeau 2018; Etrillard et al. 2019). Studies carried out in several accommodating forest power line ROW, in the USA, Sweden and France, have demonstrated their potential for sheltering varied communities of wild bees and butterflies (including rare and even reputedly extinct species) and for landscape connectivity for these insects (Russell et al. 2005; Wagner et al. 2014; Berg et al. 2016; François et al. 2018).

Potential ecological shelter bound to waterway network

Linear extent and surface area features

As regards shipping canals, most of the length (i.e. 3,175 km) is part of the network managed by the State operator (VNF), notably for heavy transport, while the remainder (428 km) is by various local authorities and mainly dedicated to recreational boating. As regards navigable rivers, the breakdown between the State operator and local authorities is more evenly balanced: 2,087 vs. 1,904 km, respectively. For control purposes, the total length of the public waterway domain was estimated with BD TOPO®: the result (6,480 km) differs by just 3% from the figure indicated by the State operator (6,700 km in Michel et al. 2015). It appears that over 80% (i.e. 5,262 km) of the VNF network length presents PES. With respect to the total 2,707 km of non-State waterway network, compiled by querying BD TOPO® and consulting network maps, more than 85% (i.e. 2,332 km) of the entire length presents PES.

Surface areas of PES bound to shipping canals and navigable rivers are divided between the public fluvial domain under State responsibility on the one hand and the non-State domain (riparian owners and local authorities) on the other. The State 68 km^2 are split between shipping canals (41 km^2) and navigable rivers (27 km^2). For the non-State domain, 25 km^2 are bound to navigable rivers and just 6 km^2 to shipping canal easements.

The total surface area of 99 km^2 is the floor value for the PES along waterway network. For navigable rivers, a strip of land can extend between the towpath and the river shore. Its width varies seasonally with the river level and may be used for cattle grazing (Etrillard et al. 2019). Similar to navigable rivers, the shipping canals ROW can also extend to the sides, at varying distance from the towpath. The national operator (VNF) indicates that the entire asset entrusted to it by the State amounts to approx. 400 km^2 (Michel et al. 2015), which covers not only the total easement surfaces calculated in this estimate (83 km^2 when urban areas are taken into account) and the aforementioned additional ROW width, but also annex areas such as the numerous former dredging sediment deposit areas, some of which have been colonized by nature (VNF 2015).

Current management of ROW

The most widespread current technique for vegetation maintenance on easements and their vicinity is grass mowing with a rotary-slasher, at least once a year (higher frequency in more anthropized areas). The common practice is to leave the cut grass on the ground. However, some experiments with pasture (sheep, cattle) have been implemented along waterway easements by means of partnerships with shepherds and farmers (Etrillard et al. 2019). For some shipping canals, banks have been vegetated to create aquatic habitat (VNF 2015).

Potential ecological shelter bound to railway network

Linear extent and surface area features

Compared to the figure provided by the national operator (SNCF) in 2015 for all operated lines (29,273 km in Michel et al. 2015), the estimate calculated from BD TOPO® for the total LGVs plus main railways (including within urban/industrial areas) differs by just 5% (30,814 km). In fact, in the span of time some main lines may have been closed and the high-speed network has increased slightly (2,141 km according to calculation vs. 2,024 km according to Michel et al. 2015). Logically then, since LGVs are designed to link distant conurbations, over 85% of high-speed network length offers PES. The ratio is similar (84%) for the extensive and highly-interconnected main railway network (total length calculated: 24,035 km). Service track is primarily located adjacent to activity centers in urban/industrial areas: consequently, just 18% of the total length in this LTI type (i.e. 8,095 km calculated from BD TOPO®) presents PES. Lastly, a large share (59%) of the total network of unused track (i.e. 4,857 km) contains PES. This part of the network typically corresponds to abandoned track located in low-density rural areas. It has been observed that disused railway lanes can foster the emergence of hedgerows, which serve as habitat and corridor for local flora and fauna (Carlier and Moran 2019). Like the rest of the railway network, unused track remains under the responsibility of the national operator, as long as these sections are not declassified. Declassification is rare and generally proceeds in the case of specific requests by a potential buyer and when it is absolutely certain that the given section will never again be used as a railway or even any other transportation purposes (e.g. bicycle path, greenway).

The total surface area of PES along railway network (333 km^2) is far lower than the figure provided by the national operator (SNCF) in 2015 for its total green areas (600 km^2 in Michel et al. 2015). The addition of the proximity strip on all interurban lines considered in this estimate (6 to 8 m depending on the line type) increases the railway side surface to 500 km^2. The gap with the operator's total figure can easily be filled by the rights-of-way located in conurbations (network estimated at 13,594 km) as well as to the sum of all rights-of-way much wider than the minimum average considered herein, e.g. when LGVs cross hilly landscapes.

Current management of ROW

The outermost part of railway ROW (i.e. the verge) is managed to maintain mixed vegetation (herbaceous and woody), with an emphasis on controlling wooded vegetation (shrubs and particularly trees) for various safety reasons (falling wood and leaves on the trafficked section, destabilization and monitoring of embankments) (SNCF Réseau 2017). Woody vegetation is generally maintained by means of mechanical brush clearing every 3–5 years. The management is more intensive on high-speed lines and leads to more grassy sides than on the rest of the railway network. Similarly to other LTIs, sheep grazing experiments are currently being conducted on grassy rail verges.

In general, unused track is left to the progressive recolonization by ruderal vegetation and this can be an opportunity for the natural reestablishment of hedgerows in some landscapes. Over time, the former verges can be replaced by trees, while the central area with old ballast offers more difficult soil conditions (rocky, dry, macro-porous), conducive for some shrubs and bushes. These wide hedgerows can provide shelter and resources to local flora and fauna (Carlier and Moran 2019).

Potential ecological shelter bound to road network

Linear extent and surface area features

For motorways, most length is managed by concessionary companies (8,088 km - i.e. 78% of the total), with the remainder (2,188 km) being by State services. Quasi-motorways, as well as dual carriageway roads are operated by public bodies, either State or local authorities. Since the motorway network is designed for long-distance interurban transportation (similar to LGVs), it is not surprising that 90% of its total length (estimated at 11,432 km from BD TOPO®) presents PES. This ratio is also high for quasi-motorways (over 75%), but shows however that for this road type, a higher fraction is located in conurbations (total length calculated from BD TOPO®: 4,597 km). As for dual carriageway roads, the fraction showing PES is 46% of total network length (i.e. 5,561 km): this reflects that an even higher fraction of this last road type is in conurbations.

For motorways, nearly 80% of the surface (129 km^2) is maintained under the responsibility of private concessionary companies, with the smaller portion (35 km^2) being under State services. The quasi-motorway sections considered in the estimate (i.e. exurban areas) are maintained by State or local authorities (i.e. department) services. Twenty years ago, the total surface of motorway verges was estimated at 160 km^2 by Meunier et al. (1998). Calculation details were not provided, but considering that in 1997 the total motorway network spanned 8,864 km (CGDD 2019), the average motorway verge width value would be 18 m. In complementing our present calculation with the two safety strips (a combined 8 m) applied to 10,276 km (Table 1) and the urban motorway network (1,156 km calculated from BD TOPO®), the total motorway verge surface amounts to 256 km^2. For the entire motorway network (11,432 km), this would suggest a theoretical average verge width of approx. 22 m, hence 4 m more than Meunier et al.'s 1998 estimated average value.

Current management of ROW

The most widespread technique for road verge vegetation maintenance is grass mowing with a rotary-slasher, generally carried out once a year as regards the outer ROW part, considered herein. Late mowing is becoming more popular among operators nowadays in order to preserve the biological cycle of flora and associated fauna. However, the cut grass left to decompose on the ground remains an obstacle to the establishment of a diverse flora and supporting insect populations (François and Le Féon 2020). For quite some time, road verges have proven to be as potentially as biodiverse as hay meadows (De Redon de Colombier 2008). The controls applied to woody vegetation are more stringent on high-speed roads ((quasi-)motorways) and generally require trees and shrubs to be situated further from the proximity strip than on classical dual carriageway roads.

Key right-of-way features with implication for ecological management

Generally, in wooded areas, power line rights-of-way are cuts made in preexisting forests. However, in some areas where the forest is expanding today due to the disappearance of pastureland, the maintenance of power line ROW keeps open parts of this former environment, such as dry grassland (François et al. 2018), particularly favorable to flora diversity and pollinating insects (Krauss et al. 2009; Jauker et al. 2013).

The broad width and virtual absence of human disturbance (limited access for people and minimum maintenance operations) for power line ROW are conducive for introducing livestock pasture, and thereby maintaining an open environment over the long term (Godeau 2018). The lateral boundary with closed to very closed environments (i.e. forest) creates a corridor linking the open environments located at both extremities or connecting other corridors. Banks of shipping canals and navigable rivers offer similar conditions for livestock. However, there is a risk of disturbance to livestock from pedestrians and cyclists using towpaths, as well as a risk of drowning due to the proximity of water (Etrillard et al. 2019).

The commercial railway network (LGVs + main railways) and road network generate a comparable surface of PES (277 km^2 and 241 km^2, respectively). The size of these ROW surfaces was estimated using the same approach, i.e. the area between the proximity/safety strip and the outer ROW boundary in a slightly rolling landscape representative of the average situation of land crossed by such infrastructure in France. These two kinds of ROW are also similar, i.e. they are distributed on both sides of a broad section traveled by high-speed vehicles. In relation to connectivity, the aim of verge management for entomofauna must not be focused on crossing traffic zones, as this would increase mortality from vehicular collision (Kasten et al. 2016; Vinchesi et al. 2018).

The particular feature of railway and road rights-of-way regarding connectivity lies in their potential to connect segments of the surrounding (and original) green network. When the LTI was built, transverse corridors of the green network were severed. As a result, on each side, the remaining segments (e.g. hedgerows) often end on the outer boundary of a uniformly inhospitable ROW. Creating supporting habitat conditions inside rights-of-way to reconnect fragments of the local green network may facilitate species flow and gene flow within the landscape: far areas (Fig. 1) can be shelters for species (e.g. simple habitat patches), then bases for dispersal of populations in the surrounding landscapes (François and Le Féon 2020). This kind of ROW can also serve as a preferential corridor for flying insects as demonstrated in the case of bumblebees foraging (Hanley and Wilkins 2015). When foraging, bumblebees and honeybees have been shown to follow flight-paths free of transverse obstacles (Ohashi and Thomson 2009; Buatois and Lihoreau 2016). Moreover, good quality flower resource favours the movement of pollinating insects within road verges, rather than across the road (Dániel-Ferreira et al. 2021). Greater floral diversity and density in the far area as opposed to the proximity strip (Fig. 1) may reduce the attractiveness of the roadside to bees, thereby reducing the risk of collision mortality (François and Le Féon 2020; Dániel-Ferreira et al. 2022).

A revised involvement for LTI operators and local stakeholders

Involvement of local skills in ecological management

Awareness of biodiversity issues related to rights-of-way is becoming widespread among LTI operators (Michel et al. 2015). However, the primary function of LTI operators is to guarantee the level of service to users. They therefore remain above all technical operators who generally lack the required expertise to implement optimal ecological management of rights-of-way. In order to address this difficulty (at least in the short and medium term), LTI staff could be accompanied by local stakeholders with skills in the management and maintenance of natural or semi-natural environments. This ecological support could be formalized through suitable management partnerships. Such local stakeholders can be NGOs of various kinds, individuals in the field of agro-ecology notably (farmers, breeders), regional natural parks or even local authorities. A small number of cooperative ventures along these lines have been, or are being, run in France; the lessons learned serve to strengthen their sustainability (Etrillard et al. 2019). These management partnerships would optimize organizational solutions combining stakeholders' legal, ecological and safety concerns. This solution helps address the wide diversity of opportunities that can arise within LTI networks across the country.

Contracting between operators and local stakeholders

According to the LTI, the partnerships involve the operator, stakeholders with environmental skills, but sometimes also the owner of the right-of-way (when the operator is not the landowner).

As regards all operated railway sections (i.e. LGVs, main railways, service track), ROW maintenance falls under the sole responsibility of the national network operator and belongs to the State-owned domain. Concerning roads, in addition to the 2,188 km of PES under the responsibility of State services (Potential ecological shelter bound to road network), 2,306 km of quasi-motorway and 1,613 km of dual carriageway roads are also counted in the State-owned domain. The other components (respectively 8,088 km, 1,175 km and 959 km) fall under the responsibility of concessionary companies or else belong to diverse local authorities (departments, local communities). As for waterways, the national operator's PES network (shipping canals + navigable rivers) extends over 5,262 km of State-owned domain while various local authorities manage the other part (2,332 km). Lastly, for power lines, the PES only extends over 711 km into State-owned forests (340 km and 371 km for HV and VHV lines, respectively), with by far the largest share located in private and municipal forests (13,007 km for HV lines + 20,281 km for VHV lines).

At the scale of Metropolitan France, the total PES along LTI networks under operations (i.e. with regular ROW maintenance), covers 85,226 km (the 2,868 km of unused railway track, generally left to be freely colonized by flora and fauna are considered separately). The State-owned domain hosts 46.2% of the operated asset. This share is made up of railways (32.0% - i.e. all the operated network), then comes 7.2% of roads, 6.2% of waterways and just 0.8% of power lines. Relative breakdown is depicted in Fig. 8. For State-owned rights-of-way, simple two-party agreements are required, between the public operator and its partner in charge of ecological management, for implementing actions to contribute to public biodiversity policies.

Non-State rights-of-way could be used in the same manner; they represent 53.8% of the total operated network with PES and develop over the domain of local authorities, private landowners or concessionary companies. This asset is mainly composed of power lines (39.1%), then roads (12.0% shared between local authorities – 2.5% and concessionary companies – 9.5%), and lastly 2.7% attributed to waterways (shared between local authorities and private landowners). For roads, in both cases, partnerships would require simple two-party agreements between the partner in charge of ecological management and the motorway concession company, or the local authority. The same would apply for waterways managed by local authorities. For power lines crossing communal or private forests, management partnerships would require a three-party agreement including the landowner (Etrillard et al. 2019).

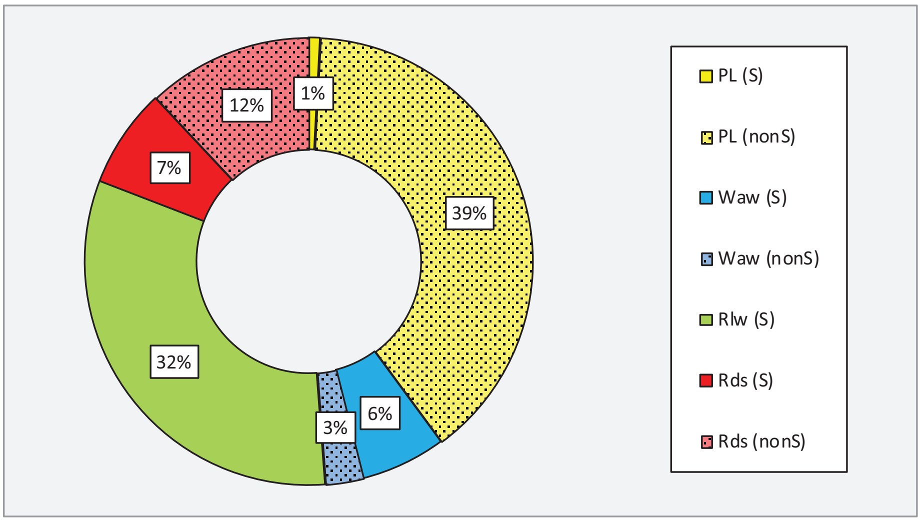

Figure 8. Share of potential ecological shelter between State-owned (S) and non-State-owned (nonS) operated networks (85,226 km). PL: power lines; Waw: waterways; Rlw: railways; Rds: roads.

Implementing national policy at local scale

The aim of the national asset estimate of PES is to provide a sound basis for an informed policy for the contribution of LTI to biodiversity conservation at national scale. Databases suitable for estimating the surface importance of PES and their ownership have been chosen in virtue of their properties of accessibility, completeness and consistency of data at broad scale. On the operational local level, the policy (local authorities) and action (field stakeholders) will require precise mapping, using more detailed data and field checking. They will take into account the specific local issues. Local mapping will enable to precisely draw the PES areas, measure their interdistances and identify their proximity to elements of the surrounding green and blue network, outside the ROW. In addition to the restoration of good habitat condition for local species within PES, this will highlight opportunities to develop stepping stones of habitat patches thanks to PES and to re-establish/develop connectivity with the surrounding landscape (François and Le Féon, 2020).

Conclusion

When a landscape has been damaged by fragmentation and destruction of its natural habitat, whether this is mainly attributable to linear transportation infrastructure or to other causes, LTI rights-of-way can, to a certain extent, provide shelter to the local flora and entomofauna through habitat and conduit functions. To optimize the use of this potential of LTI rights-of-way toward achieving a consistent and efficient conservation strategy, it is necessary to know the extent of surfaces actually suitable for developing a relevant action plan, along with the network length and spatial distribution.

Based on scientific evidence gained in recent years, ROW potential to support biodiversity is now being recognized as a viable possibility, to not only mitigate damages but also to restore local flora and fauna. However, awareness of this potential is more recent among many operators and landowners, hence ROW surface measurements in this specific aim are virtually nonexistent today. The estimate of ecological shelter potential at the national level cannot be achieved by means of simply summing local data. On the other hand, calculations based on the total length of LTI networks at the national scale would lead to gross overestimation (in terms of both length and surface area). Any kind of LTI does, in fact, present sections located in inappropriate areas for habitat and connectivity restoration, and in some cases rights-of-way are inappropriate due to their isolation or narrowness.

If cartographic databases are available at the national scale for LTI networks and land use, GIS can then provide assistance in estimating the anticipated figures, and also identify local ROW sections of interest. A GIS-based method was developed for estimating the linear extent of PES for the various types of LTI networks, as well as their respective surface areas. Calculation assumptions have been illustrated by the French context (geometric criteria of LTI cross-sections and maintenance rules for the different ROW types), but the method is adaptable and transposable to all digitally-mapped territories.

At the scale of Metropolitan France, the total linear extent of LTI networks with PES has been estimated at 88,094 km, with the largest share being for power lines (≈38%), followed by railways (≈32%), roads (≈19%) and waterways (≈9%). Each type of LTI right-of-way presents particular features that determine its ability to serve as habitat, conduit and source for the surrounding landscape, hence its ability to contribute to reconstitute green and blue networks. PES associated with railway and road networks are more evenly distributed across the country than those associated with waterways and power line networks. The minimum total surface of PES has been estimated at 2,026 km^2. While the estimates for power lines (1,353 km^2), railways (333 km^2) and roads (241 km^2) result from a sound estimate of their typical cross-sections, for waterways just a floor value based of regular easements has been calculated (i.e. 99 km^2). A specific analysis would be necessary to approximate the full PES surface of waterway networks at the national scale.

The ecological shelter potential of rights-of-way could support the conservation of common, as well as threatened flora and entomofauna. Managing rights-of-way as an ecological shelter represents a new perspective and commitment on the part of all public and private LTI operators, with implications that extend beyond their traditional core business. This perspective can also mandate new requirements and responsibilities from and for public and private landowners in pursuit of optimal ROW use. In Metropolitan France, the State is the primary landowner of PES along operated networks, which also means that over half of the assets (53.8%) fall under the responsibility of local authorities, private agents and concessionary companies. The involvement of these stakeholders could be supported in the field by means of management partnerships for suitable ROW maintenance with local actors of green and blue networks and with authorities in charge of environment and transportation, who for their part could design national or regional strategies from the figures provided by this estimation method.

The networks and total surface areas of ROW managed by LTI operators are much higher than the linear extent and surface area of potential ecological shelter. In a general way, the most biodiversity-friendly maintenance as possible must be implemented within rights-of-way. The potential ecological shelter is the most protective part of the whole asset for biodiversity with regard to disturbance factors linked to the operation of infrastructure. This is where stakeholders can engage the most ambitious and effective biodiversity conservation actions, as could be done under national/regional action plans, for wild pollinating insects notably.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

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Author contributions

François D: Environmental management, Ecology ; Medous L: Geography; GIS ; Etrillard C: Environmental law; Territorial units

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Data availability

All of the data that support the findings of this study are available in the main text.

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Research Article

Brown bear occurrence along a proposed highway route in Romania's Carpathian Mountains

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Abstract

Linear transportation infrastructure threatens terrestrial mammals by altering their habitats, creating barriers to movement and increasing mortality risk. Large carnivores are especially susceptible to the negative effects of roads due to their wide-ranging movements. Major road developments are planned or ongoing throughout the range of the Romanian brown bear (*Ursus arctos*) population, which is numerically the largest in the European Union. The planned A8 (Tîrgu Mureș–Iași–Ungheni) highway crosses the Romanian Eastern Carpathians on their entire width, posing a risk to the Romanian and broader Carpathian transboundary bear population. In the summers of 2014, 2017 and 2020, we surveyed an 80 km-long section of the planned highway using 68 hair traps with lure mounted in pairs along the route. We aimed to assess bear occurrence, genetic connectivity across the proposed highway and to estimate the minimum number and sex ratio of bears present in the area. With an effort of 3,519 hair trapping days (17 days / trap / session), we identified 24 individuals from the 45 collected hair samples, with a higher prevalence of female bears (male:female sex ratio of 1:1.3). We documented functional connectivity across the planned highway through parent-offspring (4 cases), full-sib (2 cases) and half-sib (24 cases) genetic relationships amongst sampled individuals. Terrain ruggedness and longitude were the most important predictors of bear occurrence from our analysis of detections at hair trap locations. Bears consistently occurred in the vicinity of the planned highway when in rugged terrain of the western section of the study area and were often detected close to human settlements (< 1 km). Even at this stage, without the A8 highway constructed, connectivity is likely already limited by the existing extensive network of settlements and restricted to a few important linkage areas still free of developments. Additional threats to bears and other wildlife in the area include poaching and large numbers of free-ranging dogs. We provide recommendations to mitigate these threats.

Key words: Carpații Orientali, detection survey, habitat fragmentation, hair trapping, non-invasive survey, road ecology, *Ursus arctos*

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Introduction

The loss, degradation and fragmentation of habitats represent major threats to terrestrial mammal diversity around the globe (Rands et al. 2010; Crooks et al. 2017; Kuipers et al. 2021). In recent decades, the impacts of roads on ecosystems have received concerted attention (Coffin 2007) and roads have been recognised as a main driving force behind the global alteration of natural habitats (Forman and Alexander 1998; Spellerberg 1998; Trombulak and Frissell 2000; Rhodes et al. 2014). Roads can affect many components of ecosystems and their associated edge effects can manifest at local and landscape levels (Coffin 2007). Animal species with wide ranging movements, large home ranges and long dispersal distances are especially vulnerable to roads (Rytwinski and Fahrig 2012). In particular, possible adverse effects of roads have been documented for large carnivores, including canids (Jędrzejewski et al. 2004; Riley et al. 2006), felids (Palma et al. 1999; Kerley et al. 2002; Niedziałkowska et al. 2006; Colchero et al. 2011; Litvaitis et al. 2015) and ursids (Proctor et al. 2018; Morales-González et al. 2020).

The relationship between roads and brown bears (*Ursus arctos*) is complex, because road effects can be area- and/or sex-specific, may vary by time of day and season and can be influenced by traffic volume (Penteriani et al. 2018). Roads facilitate access of people to bear habitats, increasing the chances of human-bear encounters and bear mortality risk (Benn and Herrero 2002; Ciarniello et al. 2009; McLellan 2015). In some areas, collisions with vehicles represent a major cause of documented bear mortalities (Huber et al. 1998; Kaczensky et al. 2003; Gunther et al. 2004). Certain components of roadside vegetation, especially during spring and early summer (Nielsen et al. 2004a; Roever et al. 2008a), as well as other food sources associated with human presence on roads, such as waste (Huber et al. 1998), can lure bears close or onto roads. While seasonally attractive roadside vegetation can potentially improve female body condition and reproductive success, the benefits of roadsides are countered by high mortality (Boulanger et al. 2013). Road placement, for example, in areas of low ruggedness, can combine with previously enumerated factors to further increase the attraction of roads (Roever et al. 2008a).

Some bears decrease their use of areas near roads or avoid these altogether, suggesting that roads can cause effective habitat loss at varying scales (McLellan and Shackleton 1988; Kasworm and Manley 1990; Mace et al. 1996; Waller and Servheen 2005). Displacement from habitats near roads reduces habitat extent and might affect body condition, reproductive rates and, ultimately, population density of bears (McLellan and Shackleton 1988; Mace et al. 1996). Adult males generally avoid roadsides (McLellan and Shackleton 1988; Boulanger and Stenhouse 2014). Females with cubs (McLellan and Shackleton 1988; Graham et al. 2010) and subadults (Mueller et al. 2004; Graham et al. 2010) tend to exploit the vicinity of roads more often, due to preferred forage availability (e.g. herbaceous vegetation layer for grazing) and/or as an avoidance mechanism against potentially aggressive/infanticidal adult males.

Traffic volumes are negatively correlated with road permeability for bears (Gibeau 2000; Waller and Servheen 2005; Northrup et al. 2012). High traffic levels (e.g. along highways) can create home range boundaries for resident animals (Kaczensky et al. 2003; Find'o et al. 2019). Once traffic volumes exceed a threshold of 5,000 vehicles/24 hrs, roads may become absolute barriers to bear movements (Skuban et al. 2017). Roads can also offset the social structure of bear populations, because females are less likely to cross busy roads than males (Gibeau and Heuer 1996; Waller and Servheen 2005) and stop crossing roads altogether at a lower traffic threshold than males (4,000 vs. 5,000 vehicles/24 hrs; Skuban et al. 2017).

North American studies advocate for limiting road access (Mace et al. 1996; Wielgus et al. 2002; Graves et al. 2006; Roever et al. 2008a, 2008b) and reduction of road density in bear habitat by targeted road closure and removal (Nielsen et al. 2006; Ciarniello et al. 2007; Nielsen et al. 2008; Switalski and Nelson 2011). In Europe, on the other hand, bears mostly have to contend with crowded, highly fragmented, multi-use landscapes, with little wilderness areas left (Swenson et al. 2000; van Maanen et al. 2006; Linnell et al. 2008), where they are frequently exposed to roads (Torres et al. 2016; Psaralexi et al. 2017).

Romania is an important stronghold for brown bears in Europe, hosting approximately 6,000 individuals (Swenson et al. 2000; van Maanen et al. 2006; Linnell et al. 2008; Kaczensky et al. 2013), although this number might be overestimated (Salvatori et al. 2002; Popescu et al. 2016). As a European Union (EU) Member State since 2007, Romania has plans to extend and modernise its transport infrastructure to meet EU standards, with the aid of both national and dedicated EU funding (Romanian Ministry of Transport 2008). The goal is to cope with steadily increasing traffic levels: in the period 2007–2019, the number of vehicles has almost doubled, reaching more than 8 million in 2019 (Eurostat 2021). In 2020, the total length of Romanian highways was 904 km, with a highway density of 3.8 km/1,000 km^2. Major transport infrastructure developments are envisioned to enlarge this network to a total of 2,416 km of highways and 1,784 km of express roads (Papp et al. 2022). The country's best bear habitats are in the Carpathian Mountains and their foothills (Swenson et al. 2000; van Maanen et al. 2006; Cristescu et al. 2019), with many of the planned highways intersecting bear habitat. As a result, there is a potential risk that, without proper road mitigation measures in place, the Romanian bear population and its habitats will become severely fragmented.

The planned A8 (Tîrgu Mureș–Iași–Ungheni) highway, linking the city of Tîrgu Mureș in the west to the national border between Romania and the Republic of Moldova in the east, has been identified as a major threat to brown bear habitat connectivity (Fedorca et al. 2019). In particular, its westernmost section will intersect both important bear denning habitats (Faure et al. 2020) and critical movement corridors linking denning habitats to seasonal feeding grounds, as indicated by telemetry data from bears fitted with GPS collars (Domokos, unpublished data). The goals of this study were to evaluate brown bear occurrence, habitat use and genetic connectivity along a central section of the planned A8 highway. We aimed to identify locations on the landscape that are conducive to 1) bear occurrence, 2) bear movement and 3) to estimate the minimum number and sex ratio of bears using the planned highway route prior to the highway's construction. Our overarching hypothesis was that the distribution of the brown bear population would be ubiquitous and relatively homogeneous within a landscape that maintains some permeability despite human settlements, given that highway construction had not started at the time of the study. We anticipated that specific landscape characteristics might influence local patterns of bear occurrence.

Materials and methods

Study area

The planned A8 highway is designed to traverse the Romanian Eastern Carpathians and their foothills on a west–east axis. This study covers an 80 km-long segment of the central section (Section 2) of the highway, between the villages of Ditrău in the west and Leghin in the east, representing 37.4% of the total length (Fig. 1A). Here, the planned highway will follow and upgrade an existing network of county and national roads (DJ127, DN15, DN15B). Most (approximately 55 km) of the planned highway section considered in the study also parallels human settlements, which, in Romania, are often linear when following valleys. The primary land cover in the area is forest, dominated by coniferous or mixed coniferous-broadleaf tree species, including Norway spruce (*Picea abies*), silver fir (*Abies alba*), European larch (*Larix decidua*) and European beech (*Fagus sylvatica*). Deciduous forests composed of European beech, European hornbeam (*Carpinus betulus*) and sometimes oak (*Quercus* sp.) occur infrequently mainly in some regenerating, previously logged parcels. Agriculture is mostly limited to animal husbandry and agricultural lands comprise pastures grazed from late spring to early autumn, as well as hayfields.

Even without the planned A8 highway, the study area is partially fragmented by human settlements which are contiguous in some areas. Unlike for the western section of this highway, no purpose-built wildlife crossing structures have been planned for this 80 km-long section by the Environmental Permit (Neamț County Environmental Protection Agency 2023). Instead, a series of tunnels (23), viaducts (63) and bridges (60) that were planned due to the rugged topography, were considered adequate to also function as large mammal over- or underpasses, with a cumulative length of 26.5 km (33.1% of the section of interest).

Study design

We designed a sampling scheme to quantify the occurrence, functional connectivity across the planned highway, minimum population size and sex ratio of brown bears in the vicinity of the proposed highway. Using a Geographic Information System (GIS), we divided a shapefile of the highway route into 1 km-long segments. We generated points in pairs at the end of each 1-km highway segment, with one point on either side of the highway and all points at a set 500 m from the route. Pairs of points of which at least one fell inside a GIS layer of settlements were discarded, resulting in a total of 74 points arranged in 37 pairs (Fig. 1B).

Hair trapping

We entered the coordinates of each of the 74 points described above in a handheld GPS unit and accessed the points by driving and hiking to the sites. We deployed a hair trap station at each point during three survey sessions (2014, 2017, 2020). Surveys occurred in summer (June–July in 2014, July–August in 2017 and 2020). Hair traps were active for 17 days during each survey, after which the stations were retrieved from the field. Hair-trap stations were deployed within a 50 m buffer of each predetermined point, selecting areas with trees whenever

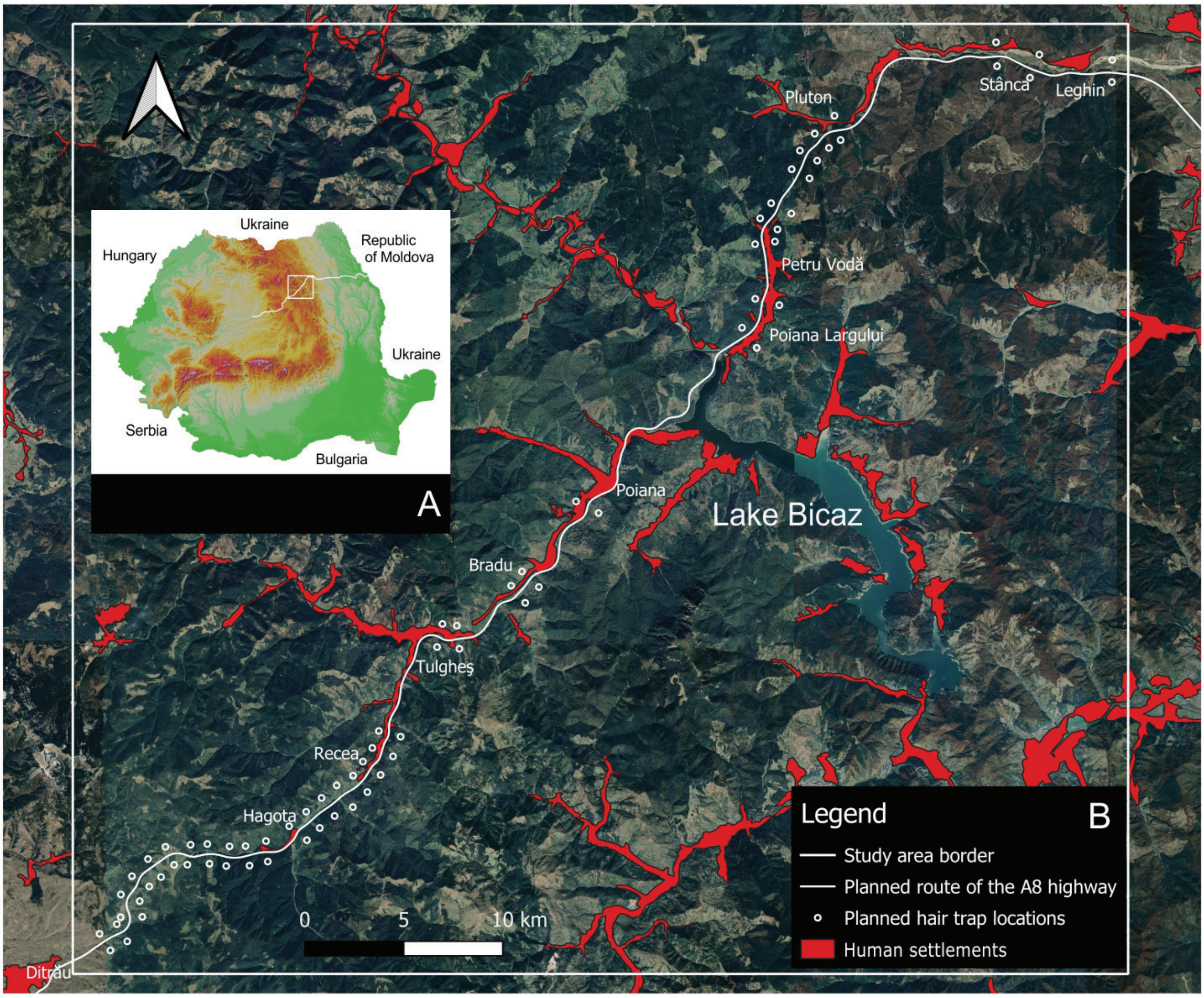

Figure 1. Route of the planned A8 highway and location of the study area in Romania's Eastern Carpathians (**A**) and detailed map of the study area, with planned brown bear hair trap locations (n = 74) situated in pairs along an 80 km-long section of the planned A8 highway (**B**).

possible. Stations consisted of a single strand of 4-prong barbed wire, mounted at a height of 50 cm that delimited a small area (6–16 m^2). The barbed wire was secured with U nails to at least three trees, if present or to 1.5 m-long, sharpened poles that we carried to the site and hammered into the ground.

At the centre of the area enclosed by the barbed wire, we constructed a small mound from locally available woody debris and rocks, onto which we poured 0.5 l of scent lure. The scented mound was unreachable for bears unless they crossed the barbed wire. We prepared the lure prior to each survey session, using 40 kg of Atlantic mackerel (*Scomber scombrus*) that we left rotting for 12 months in sealed plastic barrels. We then added 30 l of fresh, salted cattle blood and left the mixture to rot for an additional 3 months, before bottling it.

We checked the barbed wire at each station after 17 days for hair samples. Obtained hairs were visually examined to classify them as brown bear vs. other species. Examiners were experienced wildlife biologists accompanied by gamekeepers who had handled hairs of bears and other mammals, as well as physically handled bears for > 10 years. Hairs from other, clearly identifiable species were discarded after recording the non-target species. Hair samples from bears or of unclear origin were collected and labelled. Hairs located on the same group of four barbs were always collected as a single sample. Hairs located on neighbouring or almost neighbouring groups of barbs (i.e. 10–20 cm apart) were also collected as part of the same sample, unless they were obviously different in colour, length or texture. Hairs located further than 20 cm apart were always collected as separate samples, even if they seemed similar. We used medical tweezers to transfer samples from the barbed wire to envelopes, cleansing them after each use through burning with a lighter to avoid cross-sample contamination. Samples were stored individually in filter paper envelopes placed inside individual ziplock plastic bags that contained a bag of silica gel.

Genetic analysis

All pre-PCR molecular steps were conducted in a laboratory dedicated to the processing of environmental samples following standard routines for avoidance of contamination (Taberlet et al. 1999). DNA from collected hairs was extracted using the QIAamp DNA Investigator Kit (Qiagen, Hilden, Germany) with two final elution steps of 40 µl each. A part of the hypervariable domain of the mitochondrial control region (D-loop) was sequenced for general species identification (Pun et al. 2009) and haplotype assignment using the primers L15995 (Taberlet and Bouvet 1994) and H16498 (Fumagalli et al. 1996). For samples collected in the last sampling season (2020), the reverse primer was replaced through WdloopH (Caniglia et al. 2013). Obtained sequences were compared to the NCBI GenBank via BLAST search and bear haplotypes were assigned according to Frosch et al. (2014) (BG1, KJ638591.1; Ro2, X75873.1) and Matosiuk et al. (2019) (H7, MG254055.1).

For confirmed bear samples we amplified 13 unlinked autosomal microsatellite markers: Msut2 (Kitahara et al. 2000); G1A, G10C, G10P, G10D, G10L (Paetkau et al. 1995); G10H, G10J, G10U (Paetkau and Strobeck 1994); UarMU26 (Taberlet et al. 1997); Mu10, Mu23, Mu51 (Bellemain and Taberlet 2004). Reactions were performed in three multiplexes and four PCR replicates to account for genotyping errors (Navidi et al. 1992; Taberlet et al. 1999). PCR reactions were as described in Frosch et al. (2011) and microsatellite fragment analysis (including sex identification) was conducted as in Frosch et al. (2014).

The software ML-relate (Kalinowski et al. 2006) was used to infer genealogical relationships amongst individuals, based on the microsatellite data. ML-relate uses a Maximum Likelihood approach to estimate the likely relationship between pairs of individuals for four relationship categories: PO (parent-offspring), FS (full-sib), HS (half-sib) and U (unrelated). Related genotypes were manually compared to check for potential 1st grade relatives in the dataset.

Error rates for microsatellite genotyping were assessed via three basic statistics: Allelic dropout (AD) was calculated for heterozygote consensus genotypes as the proportion of one of the two consensus alleles missing across replicates (including wrong alleles); false allele rate (FA) was calculated for homozygote consensus genotypes as the proportion of additional alleles present across replicates; amplification success was calculated as the proportion of failed loci across all replicates. Error rates were calculated within samples across replicates and summarised over all samples.

Environmental covariates

We considered a suite of covariates that could *a priori* be hypothesised to influence bear occurrence (Table 1). We categorised land cover ("Habitat") in four classes which we assigned in the field when deploying hair-trap stations. We derived a terrain ruggedness index ("TRI") from a 30-m Digital Elevation Model (DEM) from the GMES RDA project (EU-DEM, https://www.eea.europa.eu/) using the GDAL Terrain Ruggedness Index algorithm in Q-GIS (v.3.18, QGIS Development Team 2013). From the resulting raster, we extracted TRI values for hair trap locations with the SAGA Add Grid/Raster Values to Points algorithm. The TRI provides a quantitative measure of topographic heterogeneity, calculating the sum change in elevation between a grid cell and its neighbouring cells (Riley et al. 1999). We calculated the distance of each hair-trap location to the nearest human settlement ("DistSett") using the GRASS v.distance algorithm. Human settlements were available as a polygon shapefile from CORINE Land Cover 2012 ("discontinuous urban fabric"; CORINE Land Cover database 2012). As human influence in the form of poaching was suspected to occur on a west to east gradient (with easternmost areas having higher poaching pressure, based on local information and our experiences in the field), we also considered a "Longitude" covariate.

Statistical analyses

We used ordinal logistic regression to investigate brown bear occurrence as a function of covariates hypothesised to influence bear habitat use. In ordinal logistic regression, the dependent variable is structured to have multiple discrete values in an assigned order. Although our data involved repeated surveys at the same set of stations, an occupancy modelling approach was not appropriate because the assumption of population closure for the survey duration was not fulfilled. Occurrence in our analytical framework took three values corresponding to

Table 1. Covariates for modelling brown bear occurrence along a planned highway in Romania's Eastern Carpathians.

Covariate	Code	Units	Data range	Linearity	Covariate justification (*potential influence to be tested in the models*)	References
Habitat						
Abiotic						
Terrain Ruggedness Index	TRI	Unitless (index)	0.43–13.86	Non-linear	Rugged terrain offers habitat security by limiting human access and providing better cover	Nielsen et al. (2004b); Martin et al. (2010); Sahlén et al. (2011)
Biotic						
Habitat	Habitat	Categorical	Pasture, mixed forest, conifer forest, deciduous forest	Non-linear	Pastures, deciduous and mixed forests provide feeding opportunities for bears. All three forest types provide cover for the species.	Dorresteijn et al. (2014); Pop et al. (2018)
Human Influence						
Longitude	Longitude	Degree	25.55–26.22	Linear	Poaching was suspected to occur on a west to east gradient, with easternmost areas having higher poaching pressure	none (area specific)
Distance to nearest settlement	DistSett	Metre	0–4,484.54	Non-linear	The proximity of settlements can filter the bear population for individuals more tolerant towards people and/or actively avoiding larger/more aggressive conspecifics	Kaczensky et al. (2006); Nellemann et al. (2007); Elfström et al. (2014)

situations where a bear was detected: no detection in the three survey sessions (1), detection in one of the three sessions (2) and detection in two or three of the three sessions (3). This method follows the approach of Chapron et al. (2014) and aims to identify areas with greatest probability of bear occurrence. Detection was defined as confirmed brown bear presence irrespective of the number of samples collected at a hair trapping station in a specific survey year and regardless of how many bear individuals were confirmed present at the site through genetic analysis.

We generated a set of 15 candidate models that were either univariate or included combinations of covariates. A correlation matrix including all covariates showed that the variables were not highly correlated (r <|0.6|) and could, therefore, be included in the same model structure. The models were included in three categories corresponding to three hypotheses: Habitat (n = 1), Human (n = 7) and combined Habitat and Human influences (n = 7). We ranked models using delta AICc and calculated evidence ratios for supported models (delta AICc < 2 and delta AICc < delta AICc of the null model).

We report the results as odds ratios, which we obtained through using the exponential of the parameter estimate(s) of the predictor(s) in the top model(s). For a one-unit increase in each predictor, odds ratios > 1 indicate an increase and odds ratios < 1 a decrease in the odds of bear occurrence.

We used QGIS v.3.16.15 for GIS procedures and R Studio v.2021.09.0 Build 351 for all statistical analyses.

Results

Overall, 68 hair-trap stations were active for 17 consecutive days across all sessions. Sampling effort was 1,156 trapping days in 2014 and 2017, respectively 1,207 trapping days in 2020. Three additional hair-trap stations active only in 2020 were excluded from modelling bear occurrence, but included in all other analyses. Three other locations that had been planned for sampling were excluded due to the presence of shepherd camps or livestock water troughs in all survey years.

Brown bear detection

During the three survey sessions, we collected a total of 89 hair samples. Mitochondrial control region sequencing was successful for 86 of the 89 analysed samples (96.6%). Half of the samples (n = 45, 50.6%) could be assigned to brown bears: 12 in 2014, 27 in 2017 and six in 2020. The samples originated from 12 hair traps in 2014 (17.7% of all mounted traps), 11 in 2017 (16.2% of all mounted traps) and five in 2020 (7% of all mounted traps). Bear hair was almost exclusively collected from hair trap locations west of Lake Bicaz (43 of the 45 samples). The two exceptions were samples collected in 2017 from the same hair trap that was the westernmost location sampled east of Lake Bicaz. During fieldwork east of the Lake, we only observed bear sign (tracks of a single animal) once in 2017. In contrast, we often encountered bear sign (tracks, scats, excavated anthills, peeled tree bark) west of Lake Bicaz.

Twenty of the 68 traps active across all three sessions registered bear hair (29.4%). Additionally, one trap active only in 2020 also captured bear hair. Fourteen traps were successful during a single session (including one active only in 2020), whereas seven traps yielded bear hair samples in two sessions each (Fig. 2A–C). Ten traps on each side of the planned highway route detected bears, whereas an additional trap only active in 2020 on the north side of the route also registered bear detection. Successful hair traps were distributed across all habitat classes surveyed: pasture (8), mixed forest (5), conifer forest (4) and deciduous forest (4).

We identified a total of three haplotypes, namely BG1 and Ro2 (Frosch et al. 2014; NCBI accession numbers KJ638591.1, X75873.1) and a third one matching to H7 (Matosiuk et al. 2019; MG254055.1, although the H7 sequence is slightly longer compared to our fragment).

Non-target species detection

We documented other wildlife and domestic species depositing hair at the hair trapping stations. Domestic dogs as the most frequently detected species overall (more than bears) were detected in all three survey years (2014, 2017, 2020: 15, 28, 22 locations, respectively) and so were wild boar (*Sus scrofa*; 2, 1, 4 locations) and red deer (*Cervus elaphus*; 1, 1, 3 locations). Roe deer (*Capreolus capreolus*) were detected in two survey years (2014, 2017: 1, 2 locations), just as cattle (2014, 2017: 2, 8 locations) and horses (2014, 2020: 1, 1 locations). Red fox (*Vulpes vulpes*; 2020: 2 locations) and sheep (2014: 1 location) were each detected in one survey year. Additionally, unidentified *Canis* sp. (either dogs or wolves [*Canis lupus*], as mitochondrial haplotypes w4, w11 and w19 following Pilot et al. (2010) have been identified in both European wolves and in dogs; data not shown) were detected in two survey years (2017, 2020: 13, 3 locations). We confirmed the presence of dogs at 10 out of the 13 locations in 2017 and all three locations in 2020 where we detected unidentified *Canis* sp. genetically.

Modelled bear occurrence

Only one model that had an intermediate number of parameters received support (model 7 with two parameters; Table 2). The model had good fit compared to the null model (Likelihood Ratio Test $LR = 14.25$, $df = 2$, $P = 0.0008$). Bears occurred consistently in areas of high terrain ruggedness. For 1-unit increase in ruggedness, the odds of bear occurrence increased by 32% (95% CI 8–63%). Longitude also influenced bear occurrence, with hair traps in the west having higher probability of occurrence than those in the east. As longitude of the trap location increased by 1-unit, the odds of bear presence decreased by 99%, with the decrease up to 98-fold as illustrated by the confidence interval (95% CI 45–9800%). Although distance to human settlement was not included in the supported model, it is noteworthy to mention that 15 of the 20 (75%) hair traps where bears were detected were located < 1 km from the edge of the nearest human settlement.

Minimum number of identified bears

Genotyping of brown bear hair samples was successful for 34 (75.6%) samples. Calculation of error rates showed a mean allelic dropout rate of 0.17 (SD = 0.28), a mean false allele rate of 0.03 (SD = 0.14) and a mean amplification

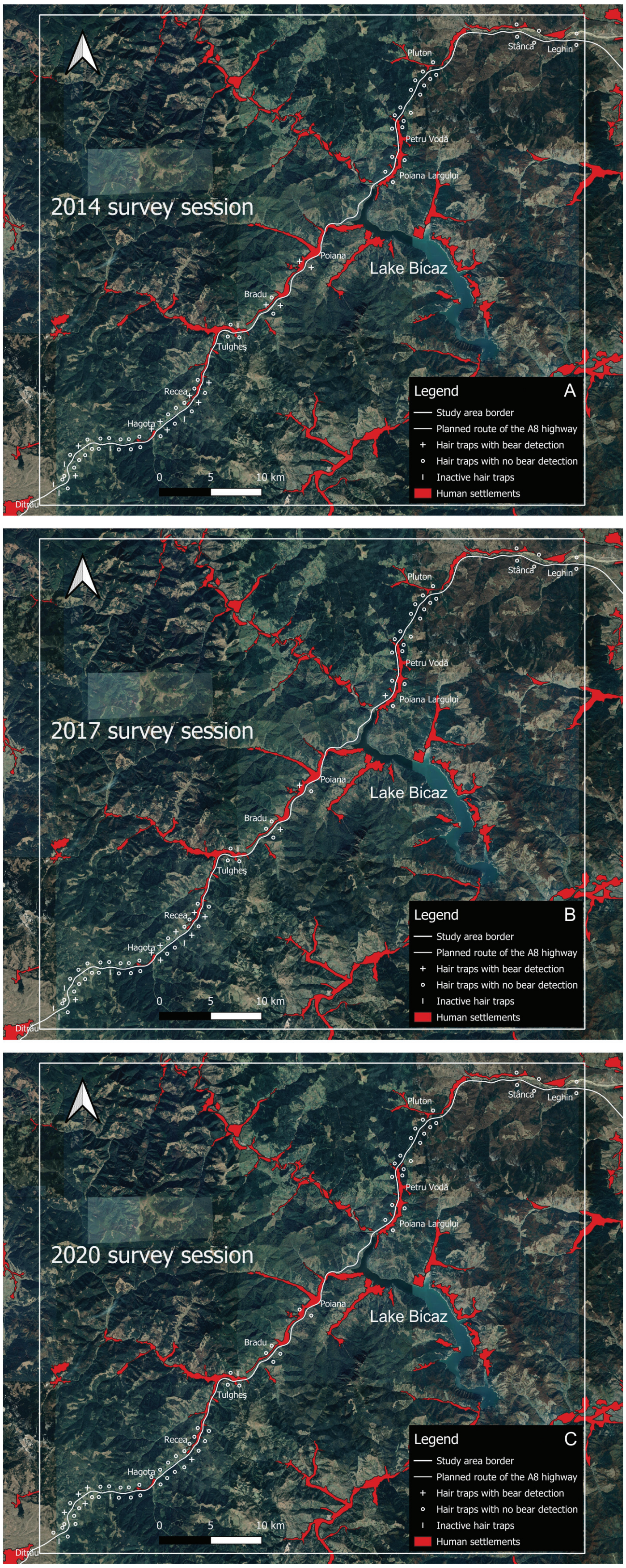

Figure 2. Brown bear hair trapping success along an 80 km-long section of the planned A8 highway during three survey sessions in the summers of 2014, 2017 and 2020 (**A–C**).

success of 0.78 (SD = 0.27). Out of the 34 samples, nine were excluded from further analysis, as they originated from four individuals that had already been identified on the same hair traps, during the same survey year (2017). We identified a total of 24 individual bears across the three survey sessions. Sex was successfully determined for 21 (87.5%) of the 24 individuals: nine were males and 12 females, resulting in a sex ratio (male:female) of 1:1.3. The largest number of individuals was identified in the year 2017 (n_{female} = 7, n_{male} = 3, $n_{unknown}$ = 2), followed by year 2014 (n_{female} = 4, n_{male} = 5) and 2020 (n_{female} = 2, n_{male} = 1, $n_{unknown}$ = 1).

Table 2. Ranking of brown bear occurrence models across a planned highway route in Romania's Eastern Carpathians. The supported model is illustrated in bold font.

Model_code	Model_set	Model_structure	K	ResDev	AIC	AICc	dAICc	ER
7	**Human Influence**	***TRI + Longitude***	**4**	**94.0**	**102.0**	**102.7**	**0.0**	**1.0**
8	Human Influence	DistSett + TRI + Longitude	5	93.9	103.9	104.8	2.2	2.9
3	Human Influence	TRI	3	100.4	106.4	106.8	4.1	8.0
6	Human Influence	DistSett + Longitude	4	98.8	106.8	107.4	4.7	10.6
5	Human Influence	DistSett + TRI	4	99.3	107.3	108.0	5.3	14.1
14	Human Influence & Habitat	Habitat + TRI + Longitude	7	92.5	106.5	108.4	5.7	17.3
4	Human Influence	Longitude	3	102.0	108.0	108.4	5.7	17.4
15	Human Influence & Habitat	Habitat + DistSett + TRI + Longitude	8	92.2	108.2	110.6	8.0	54.0
10	Human Influence & Habitat	Habitat + TRI	6	98.4	110.4	111.7	9.1	92.9
0	Null	Null	2	108.3	112.3	112.5	9.8	134.2
13	Human Influence & Habitat	Habitat + DistSett + Longitude	7	97.2	111.2	113.0	10.4	178.3
12	Human Influence & Habitat	Habitat + DistSett + TRI	7	97.6	111.6	113.5	10.8	219.1
11	Human Influence & Habitat	Habitat + Longitude	6	100.5	112.5	113.9	11.2	274.2
2	Human Influence	DistSett	3	108.3	114.3	114.7	12.0	401.0
1	Habitat	Habitat	5	107.1	117.1	118.1	15.4	2198.3
9	Human Influence & Habitat	Habitat + DistSett	6	107.1	119.1	120.4	17.8	7194.3

Bear recapture, relatedness and connectivity

Only one female bear was recaptured in our study, both within and across survey years, amongst different hair trap locations. The animal was detected in 2014 and 2017 on neighbouring hair traps located on the same side (south) of the planned highway route.

We found four cases of parent-offspring (PO) relationships and two full-sibs (FS) in the dataset (Table 3). Some potential degree of more distant relatedness (half-sibs [HS]) was identified for a total of 24 additional pairs. Eleven (9 × HS; 2 × FS) of the 30 related pairs of animals were found on opposite sides of the planned highway, which suggests gene flow across the envisioned highway route. The remaining 19 related pairs of animals (4 × PO; 15 × HS) were detected on the same sides of the planned highway (either north or south). Two of these pairs were detected on the same two hair traps. The remaining 17 pairs, however, detected on different hair traps indicate that movement is not impeded along a given side.

Table 3. Successfully genotyped bears (n = 24), relatedness and movements in relation to the planned highway route implied by detected relatedness. Hair traps A were located to the north, hair traps B to the south of the planned highway route, with numbers increasing from west to east.

Individual	Sex	Detected on hair trap (survey year)	Haplotype	Clade/ lineage	Related with (relatedness)	Movement implied by relatedness in relation to highway route
RO_UA001	♂	A06 (2014)	BG1	west	RO_UA004 (HS); RO_UA011 (HS); RO_UA022 (HS)	along; across; along
RO_UA002	♀	A13 (2014)	Ro2	east	RO_UA014 (PO); RO_UA015 (PO); RO_UA018 (HS); RO_UA021 (HS)	along; along; along; across
RO_UA003	♂	A18 (2014)	Ro2	east	RO_UA007 (FS); RO_UA009 (HS); RO_UA022 (HS)	across; across; along
RO_UA004	♀	A25 (2014)	BG1	west	RO_UA001 (HS); RO_UA011 (HS)	along; across
RO_UA005	♀	B03 (2014)	BG1	west	RO_UA006 (HS); RO_UA013 (HS); RO_UA025 (HS)	along; across; along
RO_UA006	♀	B14 (2014); B13 (2017)	BG1	west	RO_UA005 (HS); RO_UA019 (HS)	along; along
RO_UA007	♂	B18 (2014)	Ro2	east	RO_UA003 (FS); RO_UA020 (FS)	across; across
RO_UA008	♂	B20 (2014)	BG1	west	RO_UA016 (HS); RO_UA019 (HS)	across; along
RO_UA009	♂	B24 (2014)	BG1	west	RO_UA003 (HS); RO_UA019 (HS)	across; along
RO_UA011	♂	B03 (2017)	Ro2	east	RO_UA001 (HS); RO_UA004 (HS);	across; across
RO_UA012	♀	A13 (2017)	Ro2	east	RO_UA013 (PO); RO_UA022 (HS)	none (same hair trap); along
RO_UA013	?	A13 (2017)	Ro2	east	RO_UA005 (HS); RO_UA012 (PO); RO_UA015 (HS)	across; none (same hair trap); along
RO_UA014	♀	A14 (2017)	Ro2	east	RO_UA002 (PO); RO_UA016 (HS); RO_UA018 (HS)	along; along; along
RO_UA015	♀	A14 (2017)	Ro2	east	RO_UA002 (PO); RO_UA013 (HS); RO_UA018 (HS)	along; along; along
RO_UA016	♀	A25 (2017)	H7	west	RO_UA008 (HS); RO_UA014 (HS); RO_UA017 (PO)	across; along; none (same hair trap)
RO_UA017	♀	A25 (2017)	H7	west	RO_UA016 (PO); RO_UA018 (HS); RO_UA019 (HS)	none (same hair trap); along; across
RO_UA018	♂	A16 (2017)	Ro2	east	RO_UA002 (HS); RO_UA014 (HS); RO_UA015 (HS); RO_UA017 (HS)	along; along; along; along
RO_UA019	♂	B17 (2017)	BG1	west	RO_UA006 (HS); RO_UA008 (HS); RO_UA009 (HS); RO_UA017 (HS)	along; along; along; across
RO_UA020	♀	A19 (2017)	Ro2	east	RO_UA007 (FS); RO_UA021 (HS); RO_UA024 (HS)	across; across; across

Individual	Sex	Detected on hair trap (survey year)	Haplotype	Clade/ lineage	Related with (relatedness)	Movement implied by relatedness in relation to highway route
RO_UA021	?	B19 (2017)	Ro2	east	RO_UA002 (HS); RO_UA020 (HS)	across; across
RO_UA022	♀	A04 (2020)	Ro2	east	RO_UA001 (HS); RO_UA003 (HS); RO_UA012 (HS)	along; along; along
RO_UA023	?	A06 (2020)	BG1	west	–	–
RO_UA024	♂	B06 (2020)	BG1	west	RO_UA020 (HS)	across
RO_UA025	♀	B17 (2020)	Ro2	east	RO_UA005 (HS)	along

Discussion

Using non-invasive repeat survey methodology, we assessed the distribution and documented the minimum local population size of brown bears along a planned highway route in Romania, as part of an effort to collect information before highway construction. We detected bears at 21 sampling stations along the planned highway route, but with a more restricted distribution than expected and a concentration of presence in the western part of the study area. Our study did not succeed in producing direct evidence of bears crossing the planned highway route (e.g. same individual detected by hair traps on both sides of the future highway). Nevertheless, we provide genetic evidence that the population uses both sides of the planned development, including the detection of related animals on both sides of the highway route.

We found a positive association between bear occurrence and terrain ruggedness. When confronted with human disturbance, such as in human-dominated landscapes of Europe, bears may select rugged terrain (Martin et al. 2010; Dorresteijn et al. 2014; Roellig et al. 2014). Rugged terrain limits human access and provides secure habitat, minimising the risk of human-bear encounters and of human-induced bear mortalities (Nielsen et al. 2004b). With decreasing distance to human settlements, the use of increasingly rugged terrain has also been documented in the case of denning bears (Sahlén et al. 2011).

Most hair traps that detected bears were close (< 1 km) to human settlements (mean ± SD distance of all hair traps to human settlements was 1.35 ± 1.33 km). Mountainous villages in Romania commonly comprise solitary houses or small groups of homesteads, which are often not recorded as part of settlements in Corine Land Cover (i.e. discontinuous urban fabric). Thus, some hair traps that registered bears were even closer to buildings than revealed by the land-cover layer. Our results are in accordance with previous studies reporting that in Romania bears regularly use human-dominated landscapes and in general habitats in the proximity of human settlements (Dorresteijn et al. 2014; Roellig et al. 2014; Borka-Vitális et al. 2017), without necessarily coming into conflict with humans. After investigating the mechanisms underlying the occurrence of bears near settlements, Elfström et al. (2014) concluded that bears approaching settlements display a natural behaviour, best explained through avoidance of intraspecific aggression and/or interference competition. This adaptive behaviour is shaped by the despotic distribution of conspecifics more than by naivety, food conditioning or human habituation. In despotic distribution, dominant individuals exploit high quality habitats more often than subordinate conspecifics, whereas subordinate bears seem to fear dominant conspecifics more than they fear people. We can confirm that during the extended periods of fieldwork in the area, despite frequent interactions with locals, we have never heard complaints about habituated/nuisance bears, although this can be an issue elsewhere (Cristescu et al. 2016). When close to human settlements or human activity, bears may adjust their behaviour to avoid encounters with people (Ordiz et al. 2011), being most active at crepuscular or nocturnal hours to avoid overlap with human diel activity patterns (Kaczensky et al. 2006; Martin et al. 2010; Schwartz et al. 2010; Ordiz et al. 2014; Oberosler et al. 2017). Habitat selection may also vary with time of day and season according to risks associated with people, with bears near settlements selecting steep slopes and highly concealed resting sites during daylight hours (Martin et al. 2010; Ordiz et al. 2011; Cristescu et al. 2013; Skuban et al. 2018).

Longitude of the hair trap location was a good predictor of bear presence, with westernmost hair traps more successful. One possible explanation for this pattern is habitat fragmentation of the region in the west–east direction by Lake Bicaz due to its large size, as well as numerous contiguous settlements around it. However, poaching with firearms is also an issue of concern around the Lake and in the region east of it (Anonymous, Harghita County Police Inspectorate, Miercurea Ciuc, Romania, personal communication 2015, 2016). As these are some of the best bear habitats in Romania (Pop et al. 2018; Cristescu et al. 2019), widespread poaching can transform them into ecological traps for bears (e.g. attractive habitats with high mortality risk; Schlaepfer et al. 2002), also affecting bears originating from other source areas (Robertson and Hutto 2006; Lamb et al. 2017). While it is possible that some bear individuals may avoid areas with high poaching risk as an evolutionarily adaptive response to fear of humans (Ordiz et al. 2013), in general, poaching is an activity that may be difficult to predict and adapt to and could impact bear populations substantially (Kaczensky et al. 2011).

The habitat types in which the hair traps were mounted did not influence the success rate of collecting bear hair samples. In Romania, during summer when our surveys were conducted, female bears typically select mixed forests, whereas males select all three forest types: deciduous, mixed and conifer (Pop et al. 2018). Pastures are important feeding grounds for brown bears during the same period mainly because of the availability of ants (Dorresteijn et al. 2014), an important food source for the species (Swenson et al. 1999; Große et al. 2003; Roellig et al. 2014).

Although we identified 24 distinct bear individuals in the three surveys, we expected to detect a larger number of individuals. A possible reason is the close proximity of sampling stations to human settlements, which can act as a filter for the bear population, selecting for subordinate individuals or demographics of age or reproductive classes that are more tolerant towards human presence and/or actively avoid larger/more aggressive conspecifics. Nellemann et al. (2007) found that 52% of bears in the wider surroundings of settlements in Sweden were subadults of both sexes, with only 8% of adult males present in the < 10 km radius of larger settlements and resorts. While the techniques used in this study did not allow us to differentiate between age classes of bears that we sampled, we know that at least some of the detected bears were adults, as confirmed by four documented PO relationships.

Proctor et al. (2012) demonstrated that mortality associated with settlements has been a major force impacting bear populations and connectivity in western Canada, the northern United States and southeast Alaska. In our study system, we documented gene flow through PO, FS and HS relationships, both on the north–south and west–east axis and across the existing network of settlements. On the west–east axis (e.g. along the planned highway route), with the exception of some human settlement barriers near Lake Bicaz, bear movements are mostly unobstructed by human habitation, at least parallel with the planned highway route. However, because of often contiguous settlements spread along valleys stretching from west to east, bear movements on the north–south axis (e.g. across the planned highway route) are likely already limited and possibly restricted to the remaining undeveloped areas. Tunnels, viaducts and bridges that are planned for highway development could help maintain some of the remaining functional connectivity for bears and other wildlife, especially because these structures will be relatively close to each other (mean distance between structures 326.2 m (range 20–2,072 m); Silvia Borlea, EPC Environmental Consulting, Bucharest, Romania, personal communication 2023). The mean length of the planned tunnels is 188 m (range 16–940 m), while the mean Openness Index (width × height / length of the structure) of the selected viaducts and bridges is 120.6 (range 4.4–853.9). The most significant linkage areas on the north–south axis are situated between villages, such as Ditrău and Hagota (12 km), Hagota and Recea (4 km) and Petru Vodă and Pluton (4.8 km). The latter area, however, is situated east of Lake Bicaz and our study did not document bear presence in its surroundings.

Widespread, cryptic poaching could have contributed to relatively low bear detection rates in our study. Due to low densities and slow reproductive rates, large carnivores are especially vulnerable to poaching and previous studies have documented substantial effects of illegal killings on large carnivore demography (Kruckenhauser et al. 2009; Liberg et al. 2012; Persson et al. 2015; Červený et al. 2019; Benson et al. 2023). Additionally, the timing of our surveys might have also influenced bear detection rates, with part of the bear population moving during the summer to richer feeding grounds situated at lower altitudes, either to the west or to the east from the surveyed area. This pattern of significant seasonal movements of at least part of the population has been observed in another area of the Romanian Eastern Carpathians (Domokos, unpublished data) and other regions (Cozzi et al. 2016; De Angelis et al. 2021). The timing of our surveys might have influenced bear detection rates in other ways too. While our surveys took place during summer, bears are more likely to respond to scent lures in spring (Gervasi et al. 2008). Lamb et al. (2016) found that starting hair trapping at lure-scented sites towards the end of the mating season (which corresponds to early June in Romania) maximises female detections, while starting early in the mating season (late April - early May in Romania) maximises male detections. Another potential limitation of our survey design could have been the fact that scent lures do not offer a reward to the visiting bear, which might thus become trap-wise and lose interest in revisiting the site or visiting other hair-trap locations.

Even if we were unable to determine the sex of three of the 24 individual bears we identified, the data are indicative of a large population segment of females (1:1.3 [male:female]). This is comparable to the 1:1.6 sex ratio estimated for the Romanian Southern Carpathians (Skrbinšek et al. 2019) or to the 1:1.5 documented in Slovenia and 1:1.4 in Croatia (Skrbinšek et al. 2017). Prior to a ban introduced in October 2016, bear trophy hunting was a common, decades-old practice in Romania (Salvatori et al. 2002; Popescu et al. 2019). With male-biased hunting, a sex ratio skewed in favour of females is to be expected.

We showed that domestic dogs are present throughout the region, at least during the summer. The frequent detection of dogs at the hair traps is likely due to the presence of large numbers of guardian dogs accompanying livestock, stray dogs or dogs associated with human settlements. A similar finding was recorded in a study of wolf diet which revealed the importance of dogs in the diet of wolves in the south-eastern Carpathian Mountains (Sin et al. 2019). Although their benefits for protecting livestock from carnivore attacks have been demonstrated (Smith et al. 2000; van Eeden et al. 2018), domestic dogs can have negative impacts on wildlife when not under human supervision (Potgieter et al. 2016; Wierzbowska et al. 2016; Drouilly et al. 2020). If dogs as a depredation mitigation strategy are adequately applied, they can be an effective strategy for livestock protection (van Eeden et al. 2018), but may enable disease transmission at the wildlife-domestic animal interface (Borka-Vitális et al. 2017). Bears are susceptible to a number of pathogens of domestic dogs, such as canine distemper virus (CDV, the etiological agent of distemper; Di Francesco et al. 2015; Vitásková et al. 2019; Balseiro et al. 2024), canine parvovirus type 2 (CPV-2; Di Francesco et al. 2015; Vitásková et al. 2019) and canine adenovirus type-1 (CAdV-1, the etiological agent of infectious canine hepatitis; García Marín et al. 2018; Balseiro et al. 2024). Measures to decrease the risk of disease transmission between domestic dogs and bears and between livestock and wildlife in general should be incorporated into decision-making programmes for livestock husbandry under free-ranging conditions.

We also genetically confirmed the presence of unidentified *Canis* sp. (either dogs or wolves) in several locations. There is a possibility that at least some of these samples originated from wolves, in particular, the ones identified as haplotype w4, which occasionally occurs in dogs, but is commonly found in Romanian wolves (Jarausch et al. 2023). We did document wolf scats and a partially consumed livestock guardian dog in the area in spring 2015. However, our evidence indicate that most unidentified *Canis* sp. samples originated from dogs. Firstly, we confirmed the presence of dogs (either through visual inspection or genetic analysis of hair samples) at the majority of locations where unidentified *Canis* sp. samples were collected. Secondly, we collected samples later confirmed as originating from unidentified *Canis* sp. due to their resemblance to bear hair. These samples consisted of long, dark-coloured, undulating, soft guard hairs, which are also characteristic of dark-coloured, large bodied, mixed breed livestock guardian dogs. Dark-coloured dogs are traditionally used in the area, although less commonly than light-coloured animals, as shepherds appear to gradually replace mixed breed dogs with purpose-bred shepherd breeds such as Caucasian, Central Asian or Anatolian.

Conclusions

Connectivity on the north-south axis is relatively limited in the study area due to existing human settlements. Completion of the A8 highway could potentially further impede bear movements in important bear habitats centrally located in the Romanian Eastern Carpathians and their foothills. This area provides a vital link to other national-level populations located further North, including Ukraine, Slovakia and Poland (Straka et al. 2012). Together, these national populations form the vast majority of the transboundary Carpathian bear population, completed by a small population in eastern Serbia (Kaczensky et al. 2013; Chapron et al. 2014). Preserving and enhancing functional connectivity within the Carpathian bear population (Matosiuk et al. 2019; Papp et al. 2022), including maintaining permeability of Romania's Eastern Carpathians is of crucial importance (Fedorca et al. 2019). In this respect, dedicated wildlife crossing structures could have been planned by the responsible authorities during the pre-construction phase of the highway, based on the best available information concerning bear presence and movement in the area. Given the decisions already made through the environmental permit, we recommend permeability studies post-completion of the A8 highway section, with a particular focus in the area of the potential crossing structures associated with highway development as imposed by topography; and after bears have had the time to explore and start using them. If the structural features of the highway meant to bypass topographical challenges prove insufficient for wildlife connectivity, dedicated wildlife crossing structures (e.g. wide overpasses, Ford et al. 2017) might be required, even though their construction costs would be much higher at that stage. To maximise their effectiveness, these should be located in the vicinity of the still undeveloped areas identified in this study. Permanent development should be limited as movement is somewhat constricted already by the high density of human settlements, although for the time being i.e. before highway construction bear population connectivity is not yet fully curtailed. In parallel with maintaining habitat connectivity, the issues of poaching and dogs in the wild should be addressed by wildlife managers and law enforcement authorities.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

B. Cristescu and C. Domokos conceived and designed the study. C. Domokos collected the data. S. Collet and C. Nowak analyzed the genetic data. B. Cristescu and C. Domokos analyzed the occurrence data. C. Domokos led the writing of the manuscript. All authors contributed critically to drafts and approved the final draft.

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Data availability

All of the data that support the findings of this study are available in the main text.

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Research Article

A methodological framework for addressing environmental problems on aged transport infrastructure

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Abstract

While the environmental impacts of new road and motorway construction are examined in detail as part of the Environmental Impact Assessment (EIA) process, far less attention is generally paid to existing structures, some of which have been in operation for decades with no environmental assessment ever carried out. In this paper, a framework for an audit of the assessment of environmental burdens from older transport infrastructure is presented. Its main objective is to set up a systematic and comprehensive approach to the preventive identification of problematic locations on the existing road network to prepare proposals for practical and feasible upgrading or optimization measures that can be addressed within the routine repairs and small reconstructions. It primarily deals with the setup of the whole process, starting with the preparation of the background for the assessment, the field survey procedure, the design of possible measures and their subsequent monitoring. The audit concept identified a total of 14 key problem domains representing individual environmental problems, for which methodological sheets were prepared. However, this is not a rigid number; the whole framework is conceived as an open system allowing for the addition of new topics or possible methodological adaptations to the practices common in other countries or in transport sectors other than roads. The audit is currently considered as a voluntary tool applicable on the state owned transport network, thus the practical usage is in the hands of the state administration and infrastructure operators.

Key words: Auditing, biota, environmental burden, existing road network, measures, soil and water, upgrading

Research highlights

- Roads built before 1992 (in the Czech Republic) were not subject to any environmental impact assessment.
- The objective of the framework for environmental assessment of older roads is designed to highlight existing impacts needing to be addressed and the subsequent elaboration of practical optimisation measures.
- Only topics that are not part of other agendas are addressed (impact on public health, traffic safety) while the task is not just to check compliance

with limits by legislation but rather to produce recommendations to meet best practice available.

- The framework is designed as an open system, allowing for the addition of new topics or possible methodological adaptation to practices common in other countries or in transport sectors other than road.
- The audit is currently considered as a voluntary tool applicable on the state owned transport network, thus the practical usage is in the hands of the state administration and infrastructure operators.

Introduction

The issue of environmental protection in relation to transport infrastructure is a very, complex, and constantly evolving topic. The first formal Environmental Impact Assessment (EIA) system was established by the US National Environmental Policy Act in 1970 (Cashmore 2004). EU introduced Environmental Impact Assessment Directive (85/337/EEC) in 1985 and EIA was later adopted also in many other countries of the world (Petts 1999; Freitas et al. 2017), which has led to a deeper consideration of the environmental issues arising from the construction and subsequent operation of new road and motorway projects. However, despite the large infrastructure deficit typical of Central and South-Eastern European countries that is only slowly being eliminated (Rosik et al. 2018; Papp et al. 2022; Komornicki and Goliszek 2023), many existing roads were built before the EIA legislation came into force. Most of these constructions have a very long historical past and were built at a time when environmental impacts were not yet understood. The impact on environmental components was virtually not addressed at the time of their construction, or only to a very limited extent, so their real environmental impact might be much higher than if these structures were built now. These are a broad range of issues (Forman et al. 2003) that include territorial impacts (Ortega et al. 2015; Goldmann and Wessel 2020), soil erosion and drainage of contaminated water from the road (Folkeson et al. 2009; Makowska and Mazurkiewicz 2016; Rivett et al. 2016; Jandová et al. 2020), the condition of roadside vegetation (Phillips et al. 2019; Salisbury et al. 2022; Cabral et al. 2023), noise and pollution from vehicle traffic and in particular various impacts on biodiversity such as issues of ecological connectivity and the migratory permeability of roads for wildlife (Cumming and Tavares 2022; Oliveira Gonçalves et al. 2022; Papp et al. 2022), the condition of fencing and barriers for animals (Shepard et al. 2008; Beyer et al. 2014), animal-vehicle collisions (Niemi et al. 2017; Arca- Bíl et al. 2019; Rubio et al. 2023) or long-term persistence of wildlife populations (Kuehn et al. 2006; Benítez-López et al. 2010; Barbosa et al. 2020; Barrientos et al. 2021). Some authors analysed the description of the environmental problems related to transports in a holistic way, which include a life cycle approach (Erlandsson 2004) or cumulative effects assessment (Jones 2016).

When considering benefits of the transport infrastructure to the landscape, one of the key ideas for improving their landscape functions is integrating them within the system of green infrastructure (Skokanová and Slach 2020; Fňukalová et al. 2021). In recent years, emphasis has also been placed in Europe on the inclusion of green infrastructure and ecosystem services to the spatial planning

(Liquete et al. 2015; Slätmo et al. 2019; Mederly et al. 2020). Green belts along traffic roads leading to the intersection of green networks and traffic networks have great potential, especially in agricultural landscapes (Skokanová et al. 2020). The planting of grassy green belts, shrubs and trees along traffic roads can also have very positive effects on Assessment and Spatial Distribution of Urban Ecosystem Functions (Phillips et al. 2019; Nozdrovická et al. 2020; Včeláková et al. 2023).

However, unlike the EIA procedure for new constructions, the situation for the existing network is not comprehensively and systematically addressed. Only the individual sub-issues, particularly about assessing risks to public health (Fehr et al. 2014; Adamiec, Jarosz-Krzemińska 2019) or road safety (IHT 2008), are regularly addressed. At the same time, however, many sections of the older road network are reaching the end of their operational lives and significant structural or material repairs should be undertaken in the near future. This is a great opportunity to reshape infrastructure to minimalize current as well as future environmental impacts and harmonise its performance with nature conservation requirements.

Thus, there is a lack of tools that would enable a systematic evaluation of existing impacts on the older motorway and road network in relation to environmental components such as water, soil, biota, and landscape and the preparation of proposals for practical and feasible optimization measures that can be implemented within the framework of routine repairs and upgrading of roads. And it should also be a guidance for a detailed analysis of the road's environmental performance before carrying out reconstructions unless a detailed EIA is required for such a project.

The aim of the framework

The aim of this paper is to close the above gap and to propose a framework for a new tool for environmental assessment in relation to older transport infrastructure, namely the Environmental Audit of Transport Infrastructure (hereinafter referred as EADI from Czech "Environmentální audit dopravní infrastruktury"), which:

(a) complements the system of environmental impact assessment (EIA) of transport constructions, mainly focused on new constructions, by the assessment of the impact of existing roads. These were mainly built at a time when environmental protection requirements were not as high as they are now;
(b) extends the current systems for monitoring the impact of transport on public health, traffic safety and the technical inspection system for buildings to include the assessment of impacts on other environmental components, in particular water, soil, biota and landscape;
(c) is aimed at the preventive search for impact situations of the transport infrastructure and the subsequent proposal of optimisation measures to be implemented mainly in the framework of regular repairs and reconstruction;
(d) provide road managers with an overview of the ongoing optimisation measures implemented on the road network in a regularly recurring mode and provide feedback for planning and for evaluating the effectiveness of the resources spent.

Methodology

The preparation of the draft framework was carried out with regard to the legislation and the real conditions of established management practice in the Czech Republic. The initial stage of the development of the framework was a system analysis of the impact of existing roads on individual environmental components. Subsequently, from the identified environmental impacts, to avoid duplication those that are not standardised in other assessment processes were selected, while for the remaining ones the key problem areas to be addressed in the audit were defined. Through system analysis, the impact of existing roads is assessed comprehensively, taking into account the surrounding environment components. This analysis provides a holistic view of the potential consequences and allows for appropriate mitigation measures to be proposed. A separate methodology sheet was then drawn up for each key issue area, defining the treatment of the issue in five stages:

I. Introductory phase – basic analysis of the situation, screening and scoping, preparation of background documents;
II. Field survey;
III. Evaluation of results and identification of impacts;
IV. Proposal of mitigation measures;
V. Conclusion.

The last step was the elaboration of general recommendations and the proposal of a procedural course of action for the practical implementation of the audit results, including recommendations for the contracting authorities (infrastructure managers) who should further work with the audit results.

Principles and basic rules for developing an environmental audit methodology

When designing and developing the framework, we applied the following rules as much as possible:

1. Focus on current impacts reduction and prevention of long term effects – the goal is to find problematic impact sites so as to solve reduce environmental consequences. Therefore, it is proposed to carry out a comprehensive assessment for each sub-section of road and to include this tool as a regular part of roads' management.
2. Independence from legislative requirements – the aim is not just to check compliance with limits by legislation but rather to meet best practice available.
3. Practical orientation – the audit is based on the assessment of selected situations and its outputs are focused on realistic mitigation measures that can be implemented mostly within the framework of routine maintenance or basic reconstruction of the road. The environmental audit is in no way analogous to the EIA process; it does not seek a theoretical

description of possible impacts and synergies, but rather a search for practical proposals to mitigate traffic impacts.
4. Efficiency of processing – an environmental audit fundamentally avoids duplication of assessment. It therefore does not include procedures that are already regularly implemented today.
5. Respect for local conditions – given the considerable variability of both roads and surrounding natural conditions, the EADI is designed as an open system whose basic methodological approaches must always be adapted to the specific local situation. Therefore, at the beginning of each EADI, a screening and assessment of the local situation should be carried out and the scope and methodology of the assessment modified accordingly.
6. Spatial extent – the audit focuses on the situation in the open landscape and thus it is determined that the EADI is not intended to assess the situation in the urban areas.

Systematic analysis of the environmental impact of existing roads

The initial stage of the framework development was a systemic analysis of the impact of existing roads on individual environmental components. For defining the EADI framework, a matrix was proposed:

- The vectors that transmit the impact of road on the components of the environment (see Table 1). The advantage of vectors is that they are in most cases quantifiable (noise, emissions, concentrations of substances in water, etc.).
- Environmental components according to the EIA outline – see Table 2.

The individual impacts identified (Table 3) were investigated in terms of their inclusion in the already regularly ongoing agendas of various government bodies and road managers. In order to avoid duplication and to maxi-

Table 1. Vectors transmitting the impact of road (physical infrastructure + traffic) on the various components of the environment.

Vector		Description
A1	mechanical motion energy	traffic accidents, animal-vehicle collisions, wildlife mortality
A2	acoustic energy	noise, noise disturbance of inhabitants and wildlife
A3	light (electromagnetic energy)	light pollution of the environment
A4	transport of airborne substances	dispersion of emissions
A5	transport of waterborne substances	contamination by (a) substances from winter maintenance (b) other substances from traffic
A6	visual perception	disturbance of the human and animal population by the movement of motor vehicles; disturbance of the landscape character
A7	barrier effect	(a) physical, (b) psychological barriers to wildlife and human movement
A8	modified habitats	change in microclimate, distribution of plants and animals; change in land-use and landscape matrix

mise the efficiency of the process, several sub-issues were not included in the EADI, because there is an established monitoring framework for them. Such sub-issues are:

- public health impact assessment,
- road safety assessment,
- assessment of the technical condition of structures on road.

EADI is thus focused on impacts that are not yet systematically monitored although they may be in relation to some of the topics listed above (e.g. road safety assessment with the Animal-Vehicle Collisions). These have been clustered into three areas:

- biota,
- soil and water,
- landscape and cultural heritage.

Table 2. Environmental components according to EIA legislation (Czech law act no. 100/2001 Coll.).

Environmental component		Description
B1	inhabitants	population and public health
B2	atmosphere	atmosphere and climate
B3	noise	noise situation and other physical and biological disturbances (light, vibrations)
B4	water	surface water and groundwater
B5	soil	soil cover
B6	natural resources	natural resources
B7	biota	biodiversity, fauna, flora, ecosystems
B8	landscape	the landscape and its ecological functions
B9	immovable property	immovable property, cultural heritage, architectural and archaeological monuments

Table 3. System analysis of major environmental impacts of roads – matrix of vectors (A) and environmental components (B).

		A1	A2	A3	A4	A5	A6	A7	A8
		Kinetic energy	**Acoustic energy**	**Light**	**Airborne movement**	**Waterborne movement**	**Visual contact**	**Barrier effect**	**Biotope change**
B1	**inhabitants**	traffic accidents	noise disturbance	lighting pollution	imissions	pollution	disturbance	fragmentation	
B2	**air**				emission				
B3	**noise**		noise disturbance						
B4	**water**					pollution			
B5	**soil**				emissions	pollution erosion			
B6	**resources**								
B7	**biota**	collisions mortality	noise disturbance	light pollution	imissions	pollution	disturbance	fragmentation	spread of species
B8	**landscape**		noise disturbance				landscape character	fragmentation	land-use
B9	**property**	direct damage			corrosion of materials	corrosion of materials	landscape character		

Key problem domains

Based on the authors' long-term practical experience and according to the numerous literatures (e.g. Hlaváč et al. 2020; Adamec et al. 2008; Rodrigue 2020; van der Ree et al. 2015), the most serious and frequent risk factors in relation to transport were identified for each EADI area. These are further identified as key problem domains (KPD) and form the methodological basis for the assessment. A list of the individual KPDs which have been addressed in depth in the methodology is presented in Table 4.

Each KPD has defined its own methodological procedure, which is based on the practice standardised for the individual domain addressed. The methodological procedures for individual KPDs are described in detail in the EADI certified methodology (Dostál et al. 2021), a sample of such a methodological sheet is presented in Appendix 1.

The list of KPDs above may not be fixed, EADI is designed as an open system. KPDs form the basis of the assessment and in the EADI they must be assessed compulsorily on all domains from Table 5 despite some the domains may be identified as non-relevant for the assessed section of road. If a different problem domain (e.g., invasive plant species) occurs on any of the assessed sections, it will be either specified directly by the contracting authority or identified in the screening process and included in the assessment.

Each KPD:

- Represents a clear practical problem that requires a concrete practical solution in several places. This is based on the practical focus of the EADI to ensure that realistic optimisation measures are associated with each factor assessed.
- Is a kind of coherent issue with its own methodology and scientific literature as shown in Appendix 1. The recommendations from this literature then also serve as a basis for the design of the measures.
- Is described according to uniform scheme (see Table 5), which is binding in the EADI. If another KPD is added as part of the screening, it will also follow this outline.

Table 4. Set of key problem domains (KPD).

EADI area		Key problem domain		Impact
i	biota	B1	Permeability for large mammals	fragmentation, land-use
		B2	Traffic accidents with wildlife	mortality
		B3	Critical sites for amphibians	mortality, fragmentation
		B4	Migration along watercourses	spread of species, fragmentation
		B5	Concept of fencing	mortality, fragmentation
		B6	Noise protection walls	noise disturbance
		B7	Impacts on small special protection areas	disturbance, spread of species, land-use
ii	soil and water	V1	Road drainage concept	pollution, mortality
		V2	Winter maintenance technology	pollution
		V3	Watercourse fragmentation	spread of species, fragmentation
		V4	Slope instability and landslides	erosion
iii	landscape and cultural heritage	K1	Visual disturbance and landscape character	landscape character
		K2	Accessibility and permeability of the landscape for inhabitants	fragmentation
		K3	Impacts on immovable cultural heritage	direct damage, corrosion of materials

Table 5. Binding scheme of each KPD.

Id	Heading	Description
A	Name	KPD working title
B	Component of Environment	classification within environmental components and subcomponents
C	Characteristics	basic description of the problem area, reasons for the solution
D	Background materials	baseline documents, input for the field investigation
E	Field survey	field survey procedures, monitoring considerations
F	Evaluation criteria	methodological criteria for determining the correct solution
G	Proposal of measures	basic conceptual design for the implementation of practical measures
H	Summary	conclusion for the evaluation in the domain

Workflow for audit processing

Formally, the preparation of the EADI is divided into 5 basic phases. The methodological procedure for each stage is strongly dependent on the environmental problem addressed and is therefore defined within the framework of the methodologies established for the processing of individual KPDs. However, a set of general recommendations can also be established that apply to the individual stages.

I. Introductory phase – basic analysis of the issue, screening, and scoping

- The screening and scoping phase allows for the adaptation of the methodology of work on individual KPDs to specific local situation and ensures variability in the overall approach.
- Based on the input data and other available information, a decision will be made on the possible extension of the assessment to other environmental components (additional KPDs beyond those listed in Table 5).
- In relation to the specific situation, the level of detail of the assessment will be chosen for individual KPDs in relation to individual road features. Depending on the specific assignment, it is possible to evaluate (i) only the objects that are directly related to the KPD in question (this is the basic solution), or (ii) to evaluate all objects of the section (variant solution – where the survey is focused more on the theoretical level) or a combination of both approaches can be used where desirable.
- As a basis for the subsequent field survey, sufficient data should be obtained on the road to be assessed, its immediate surroundings and, if necessary, on the immediately adjacent sections of adjacent roads. The structure of the required data varies from one KPD to another.
- Geographical information systems (GIS) shall be used as much as possible in the data processing.

II. Field survey

- The methodology for the field survey is determined individually for each KPD.
- Common to all field surveys is the need to accurately identify the objects to be assessed on the road. Two procedures are recommended to be applied simultaneously, using: (a) geographical coordinates (most often in

the WGS-84 system), allowing for general use in GIS; (b) positioning (in km) of the road in question – commonly used by road managers.

- Each object evaluated during the field survey is described using predefined forms, which are specified within the individual KPD method sheets – for an example see Appendix 1: Table A1 and completed form in Appendix 2.
- Local knowledge can also provide valuable information, so it is desirable to consider information obtained, from local stakeholders such as hunting associations, municipal governments, locally competent forest or water stream managers and residents.
- Basic photo documentation shall be taken during the assessment, preferably with equipment that allows geographical coordinates to be included to the meta-information of the image taken.

III. Evaluation of results and risk identification

- The guidance for this stage is based entirely on the procedures described in the individual KPD method sheets. It includes an assessment of the existing impacts and identification of the risk of impacts that may occur in the long-term.

IV. Proposal of mitigation measures

- The measures proposed are mainly of recommendatory nature. It contains the basic type of measure and initial specification (e.g., a two-sided fence on the road at km x-y with a fence height of 1.5 m). The proposals are not developed into design details.
- The proposal may also include a recommendation for further detailed investigation to clarify the issue and provide the necessary information for the final decision (e.g., chemical analysis of water from the road, detailed monitoring of animal movements, etc.)
- Detailed design and implementation of the measures shall be carried out after the audit.
- The individual proposed measures can be classified in terms of the possible implementation horizon as short-term (e.g., low-cost measures such as the addition of traffic signs, minor vegetation improvements, etc.), mid-term (e.g., speed limitation through physical measures, planting of guiding greenery, barriers for amphibians), and long-term (investment-based actions such as the construction of an ecoduct).

V. Conclusion – recap

- It is used for a comprehensive overview of the section and the proposed measures.
- A basic formal evaluation statement for each KPD is proposed:
  0 – the impact does not occur on the evaluated road,
  1A – the impact is present on the road and the current state meets the requirements of environmental protection,
  1B – the impact occurs on the road and the current condition is unsatisfactory – a list of noncompliant sites and an overview of the proposed measures are given.

Recommendations and discussion

The framework presented for EADI is a fully voluntary tool that does not introduce any formal approval processes. The EADI is designed as a methodological tool, which should be implemented by road managers to get a better and timely overview of environmental problems in the managed section of an existing road. It provides summary information on the environmental impacts of the road in question; it presents a list of optimisation measures as a basis for their further refinement and elaboration (more detailed studies, monitoring, project preparation, economic analyses, etc.). This is a recommended base for the preparation of investment plans for reconstruction and repair of the road sections in question; preparation for changes in maintenance technologies; summarisation of data on the optimisation measures implemented so far and their effectiveness as a basis for optimising the use of financial resources. EADI can be a suitable basis for the development of project documentation for the upcoming road reconstruction. It is assumed that in the most common cases the contracting authority and initiator of the audit will be the manager or owner of the road in question. The process of setting up the methodological framework itself makes maximum use of verified approaches and available information to avoid increasing the workload and financial demands of the whole process and duplication with other activities.

The basic aspects for setting priorities when assigning the sections to be subjected to environmental audit are upcoming road reconstruction; changes in maintenance technology; identified problems in environmental protection; immediate contact of the road with environmentally sensitive areas, or suggestions by state administration authorities or citizens. The implementation of the EADI is to be carried out only by professionally competent entities with practical experience in the field of environmental impact of road infrastructure.

As a new tool only finalised and approved at the end of 2021, the EADI is currently in the process of raising awareness of its existence and potential benefits. So far (summer 2023) full audits have been carried out on three sections of the road network (one each on a motorway, a national road and a Class II Road). Partial audits (consisting of 3 selected KPDs focusing on landscape fragmentation) have been prepared for the post-project evaluation of two national road constructions implemented under the Operational Programme Transport. The practical implementation of the individual recommendations from the audits is a subsequent task, which is the exclusive responsibility of the individual administrators of the evaluated infrastructure.

The proposal for a new environmental auditing tool is designed for use in the road transport sector. However, it is conceptualised in such a way that its analogy for other types of linear infrastructure is possible. But the specific problems of each mode of transport must be considered from the very beginning when creating such an analogy. Focus on the relevance of the individual KPDs and subsequently adapt their methodological sheets on the basis of the practices and methodologies used in the individual transport sectors. The similar situation is also with the regional transferability. Our methodology is adapted to the situation in the Czech Republic, but the basic concept of audit can be used in other countries as well. In this case, it will be necessary to consider relevancy of specific problems in each region by the selection of individual KPDs and to revise their methodological part to the standards required in each country.

Conclusion

Ageing infrastructure is a global problem with potentially harmful consequences to the environment and innovative approaches are required to address this. The proposed framework provides a comprehensive and integrated approach to assessing the impacts of existing transport infrastructure on the surrounding environment. It sets up a step by step process, starting with the preparation of the background information for the assessment, the field survey procedure, the design of possible mitigation measures and their subsequent monitoring. Thus, EADI proposes a systematic approach to the preventive search for problem sites on the existing motorway and road network in relation to hitherto less monitored environmental components such as water, soil, biota, and landscape, with the aim of preparing proposals for practical and feasible optimisation measures that can be implemented primarily in the context of routine road repairs and reconstruction. Authors are convinced that EADI has the potential to introduce a systematic approach to assessing the impact of existing roads on the environment. It could also find its use in the post-project evaluation of newly constructed roads if the audit is extended to check compliance with the conditions set during the EIA process.

Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Data availability

All of the data that support the findings of this study are available in the main text.

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Appendix 1

Sample KPD methodological sheet – Permeability for Large Mammals

KPD B1: Permeability for Large Mammals

A. Name

Permeability for Large Mammals.

B. Component of Environment

Biota – fauna – mammals – population fragmentation.

C. Description

The sensitivity of different species to the barrier effect of roads varies. The most sensitive species are those that inhabit large ranges and the interconnection of sometimes strongly separated sub-populations is necessary to maintain their long-term existence. These animals include species collectively referred to as 'large mammals'. In the Czech Republic these include the brown bear (*Ursus arctos* L.), grey wolf (*Canis lupus* L.), the Eurasian lynx (*Lynx lynx* L.) and European elk (*Alces alces* L.) These species are protected under Act No. 114/1992 Coll. on the Protection of Nature and Landscape as specially protected species. Given the long-distance nature of migration, the protection of migration corridors must be addressed conceptually at national level.

For this purpose, the Agency for Nature Conservation and Landscape Protection of the Czech Republic has defined the so-called "Habitat of selected specially protected species of large mammals" (Romportl et al. 2017) as a protection tool. The Habitat includes areas allowing breeding and long term stay (so-called core areas) and migration corridors used for movement between the core areas. The map layer is provided in the system of spatial planning as a Planning Analytic Material (ÚAP), phenomenon 36b (AOPK 2023). Given that the ÚAP phenomenon 36b has the character of a relatively dense closed network with a total area of more than 25% of the Czech Republic, it is logical that motorway and road constructions must come into contact with this layer, and it is therefore necessary to address these conflicts. Each contact with this network must be assessed separately.

The assessment must be carried out on all motorways, expressways and other more than two-lane or fenced Class I roads. Class II and III roads and two-lane unfenced Class I roads are generally considered to be passable and do not require any construction measures. Nevertheless, it is recommended that even on these roads the contact points should be inspected in the field and assessed for any barrier.

D. Background materials

- map layer "Habitat of selected specially protected species of large mammals" – ÚAP phenomenon 36b.
- working map – contacts of the assessed road with the ÚAP phenomenon 36b.

Table A1. Migration object assessment form.

General description			
Object id:	Positioning on road [km]:		Type of object: • underpass/overpass • type category according to Hlaváč et al (2020)
Landscape element	Most of the primarily designed structures lead another landscape element (e.g., dirt road, forest road, road, watercourse, etc.) across the road.		
map of the surrounding area			
Ecological characteristic of the site			
Importance of migration route	Description of migration routes at supra-regional, regional and local scales. Description of elements of the territorial system of ecological stability supporting animal migration.		
Landscape structure – supporting effects	It is a description of the elements that support migration, create migration pressure and increase the likelihood that the pathway will be used. These include the presence of a watercourse, valleys as natural migration routes for animals, the presence of green areas, food supply, etc.		
Disturbations	Description of the components that obstruct wildlife migration, create migration resistance, and reduce the likelihood that the migratory path will be used. These include the presence of transport, industry, mining, agricultural activities, proximity to settlements, etc.		
Technical characteristics of the site			
Type of wildlife passage	Description of the migration object given by the engineering design		
	Width (m):	Height (m):	Length (m):
Sub-bridge surface type	The nature of the surface must be natural as much as possible, the most suitable surface is grassed, natural soil without vegetation is also possible, paved concrete or asphalt surfaces, gravel, pebbles are completely inappropriate. Other potential disturbances associated with the sub-bridge should be mentioned.		
Shelters	The aim of the shelters is to compartmentalize the migratory object, to provide shelters for small animals and to facilitate their movement around the object (e.g., logs, branches, stones, etc.).		
Waterstream	The method of waterflow diversion determines whether the object will be usable for migration in addition to its hydrological function. As far as possible, it is advisable to leave the stream in its natural state and to leave a dry path, preferably along both banks of the stream.		
Surroundings			
Fencing	Wildlife that encounters road fencing often follow the fence and can be led to a migratory object. Fencing should be implemented on both sides of the object always from a migratory object to the next. Free endings of the fence without connections to migration objects are not recommended.		
Guidance vegetation	The aim is to guide the animals to the object.		
Summary			
Overall narrative evaluation of the migration object. Note on the proposed modifications to the object.			

E. Field survey

On the basis of the prepared background materials, a field survey will be carried out, including photographic documentation. There are two basic types of contacts that can occur in the working map:

a) corridor crossing – each crossing site is surveyed separately. Migration objects that meet the requirements for large mammals (animals of A category according to methodology Hlaváč et al. 2020) shall be described according to the outline in the Appendix 1: Table A1 above.
b) core area passage (contact section) – all migration objects that meet the parameters for Category A throughout the contact section shall be evaluated. This is a less common case, as core areas are mostly located in national parks and protected landscape areas where motorways and other capacity roads are rare. Distribution of wildlife passages (overall number and distance from each other) will be of the utmost importance for assessment.

F. Evaluation criteria

Basic concept for evaluation is that Class II and III roads and two-lane Class I roads – unless fenced or equipped with another impassable barrier – are not considered impassable migration barriers and no special migration facilities need to be implemented. On the other hand, motorways, three- and multi-lane Class I roads – and fenced roads of all classes – shall be considered an impassable barrier and suitable large mammal migration objects (Category A according to Hlaváč et al. 2020) shall be provided at the intersection with the migration corridor to ensure connectivity for wildlife.

Evaluation of the permeability (suitability) of the migration object:

- the passage must meet both technical and ecological parameters;
- minimal dimensions of the passage are set-up by Hlaváč et al. (2020):
  (i) overpass – central width min. 40 m;
  (ii) underpass – width 20–40 m, height 5–10 m, openness index 5–8;
- other technical parameters – nature of the underbridges, continuity of fencing, vegetation arrangements etc. – see comments in the Appendix 1: Table A1;
- basic ecological requirements:
  (i) minimisation of disturbing elements to avoid blocking the migration route;
  (ii) sub-measures – see Appendix 1: Table A1;
- parameters for contact sections (passage through the core area) – suitable wildlife passages for large mammals should be separated by a maximum of 5 km. This is an indicative figure; the whole situation needs to be assessed individually.

In most cases, the suitability or unsuitability of a migration object can be decided during a field survey based on the above criteria. In borderline and controversial cases, the literature should be consulted.

G. Proposed measures

It is dictated by nature conservation legislation that the migration passage for large mammals (as determined by ÚAP, phenomenon 36b) must be ensured. Therefore, if there are no suitable migratory features on the assessed road at the point of contact, they should be proposed.

Within the EADI, the proposal of new wildlife passage is conceptual and delineates only:

a) type of construction: partial modification of an existing facility or construction of a new facility;
b) location;
c) construction type (overpass/underpass) – categorisation according to the methodology by Hlaváč et al. (2020);
d) basic dimensions;
e) timeliness of the solution.

For the final design, it is advisable to prepare a detailed migration study before starting the investment preparation.

H. Conclusion

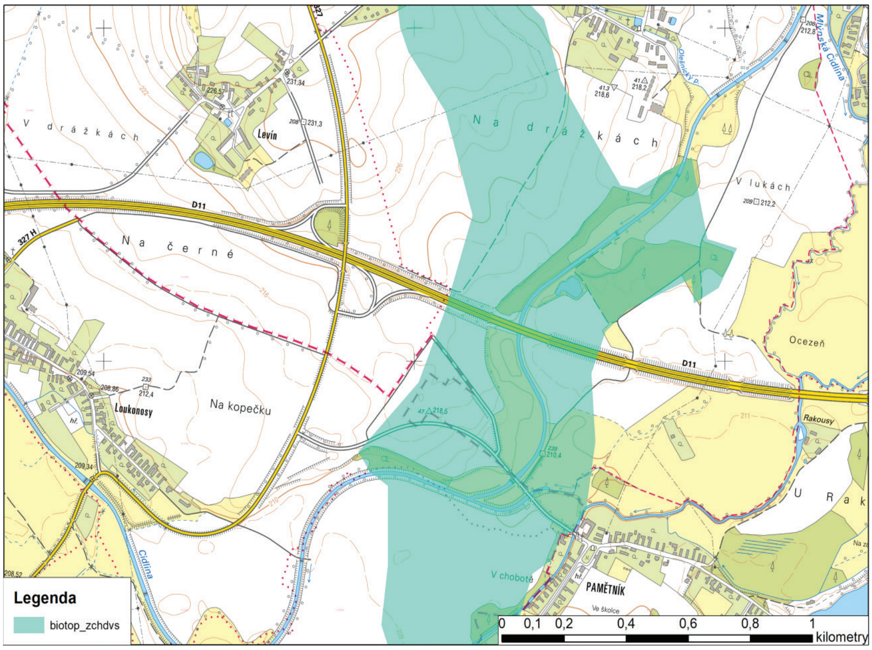
The result is a decision between three alternatives:

0 – the impact does not occur on the evaluated road,
1A – the impact is present on the road and the current state meets the requirements of environmental protection,
1B – the impact occurs on the road and the current condition is unsatisfactory – a list of noncompliant sites and an overview of the proposed measures are given.

Appendix 2

Table A2. Example of completed migration object assessment form.

General description			
Object id: D11-066	Positioning on road [km]: 62.2		Type of object: underpass P6
Landscape element	Watercourse – river Cidlina		
map of the surrounding area			



Ecological characteristic of the site			
Importance of the migration route	This is a migration profile of supra-regional importance. The D11 motorway crosses the Biotope of selected specially protected species of large mammals. Furthermore, the regional biocorridor of the territorial system of ecological stability is designed under the bridge.		
Landscape structure – supporting effects	Along the Cidlina river there is a shrubbery in some places, to the north of the crossing point the mature trees turn into a small forest. Approximately 1.5 km from the building, a corridor leads through a larger forest in the south and north.		
Disturbations	The villages of Olešnice and Pamětník are more than 500 m away from the migration object.		
Technical characteristics of the site			
Type of wildlife passage	Wide bridge across the riparian floodplain on the lower reaches of the river		
	Width [m]: 200	Height [m]: 3.5	Length [m]: 30
Sub-bridge surface type	Natural clay surface of the underbridge. The Cidlina river is reinforced with stone under the bridge.		
Shelters for small animals	Shelters are not present.		
Waterstream	The Cidlina river is reinforced with stone under the bridge.		
Surroundings			
Fencing	Fencing is implemented on both sides, there is an incorrect design of the fence ending – leaving approximately 20 m gap between the end of the fence and the bridge railing		
Guidance vegetation	Guidance vegetation is not present.		
Summary			
An excellent, sufficiently dimensioned migration object for all categories of animals. Fence maintenance and technical solutions are important, it is necessary to complete the fencing so that it is properly attached to the bridge			

Figure A1. Migration Object nr. D11-066 north-west from Pamětník.

Figure A2. Inappropriate gap between the end of fencing and migration object nr. D11-066.

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Research Article

Traffic mortality of wild forest reindeer *Rangifer tarandus fennicus* in Finland

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Abstract

Vehicle collisions kill at least one million ungulates annually in Europe. The number of traffic-killed individuals is usually relatively low for managed species, compared to the annual harvest quota. Therefore, traffic mortality in common ungulate species has historically been seen as a management and traffic safety problem, rather than a conservation issue. However, rare ungulate species, such as European wild forest reindeer (WFR) *Rangifer tarandus fennicus*, challenge this paradigm. The global population of WFR is approximately 5 300 individuals, divided into three main subpopulations: Suomenselkä and Kainuu in Finland, and N-W Russia. WFR females generally produce only one calf per year, which makes this species particularly vulnerable to any additional source of mortality. Here, we investigate traffic mortality of WFR in Finland. For both Finnish WFR subpopulations we estimated a kill rate (the proportion of individuals killed/struck) and, in relation to their winter population sizes, the collision and traffic mortality rates. Our collision data was collected during 2017–2022 by volunteer hunters and consisted of 390 road traffic collisions (407 WFR individuals), with supplementary data on railway collisions. In total, 259 individuals were killed directly in road traffic collisions or euthanized later after tracking (kill rate 64%). An age class (adult/juvenile) was determined for 265 animals (65%), and the results indicated that noticeably more adults than juveniles were killed. In relation to wintering subpopulation sizes, there were higher collision and traffic mortality rates in Suomenselkä (3.0% and 2.0% of the winter population, respectively) than in Kainuu (1.8% and 1.3%). WFR-train collisions occurred in both subpopulations. In Suomenselkä, a railway mortality rate of 0.2% was recorded, while in Kainuu it was 0.7%. We found collision and traffic mortality rates that were relatively low and comparable with those of other ungulate species. However, the relatively high proportion of adults observed among road-killed individuals lends support for further studies to develop species-specific mitigation measures for WFR.

Key words: Collision rate, road-kill, traffic mortality rate, ungulate-vehicle collision

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Introduction

It is estimated that almost 30 million mammals are killed on European roads annually (Grilo et al. 2020), with ungulates accounting for at least one million of those deaths (Langbein et al. 2011). During the last few decades, an increase in certain ungulate populations has led to an increased number of ungulate-vehicle collisions (UVCs) (reviewed by Valente et al. 2020). Most European ungulate species that are recorded in collision statistics are managed by hunting, e.g., wild boar *Sus scrofa* and roe deer *Capreolus capreolus* (Linnell et al. 2020), and the annual number of traffic-killed individuals is usually relatively low when compared to the annual harvest quota (Seiler et al. 2004; Niemi et al. 2015; Neumann et al. 2020). Traffic mortality has therefore primarily been seen as a management and traffic safety issue, especially for species which are overabundant (reviewed by Carpio et al. 2021).

While collisions with common and abundant ungulate species are seen mainly as a traffic safety issue, traffic mortality of some endangered species or isolated populations can reach such high rates as to negatively affect population levels. A well-known example is Florida key deer *Odocoileus virginianus clavium*; approximately half of its documented mortality was due to traffic before mitigation measures were implemented in the riskiest road sections (Lopez et al. 2003; Parker et al. 2011). Dekker (2021) found that traffic mortality could partly explain the decline of elk *Cervus canadensis* and bighorn sheep *Ovis canadensis* populations in Jasper National Park in Alberta, Canada, and Hegel and Russel (2013) suggested that road mortalities could become a future conservation concern for mountain caribou *Rangifer tarandus caribou* in Yukon, Canada.

The persistence of many wild Rangifers is threatened by several anthropogenic factors, such as climate warming, landscape change and traffic-related mortality (Vors and Boyce 2009), and many populations or herds of caribou and reindeer have declined across their range (Gunn et al. 2009). One ungulate species which is potentially negatively affected by traffic mortality is the European wild forest reindeer (or Finnish wild forest reindeer; WFR) *Rangifer tarandus fennicus*, a rare subspecies of the circumpolar reindeer *Rangifer tarandus*. Its conservation status is listed as near threatened, according to the 2019 Red List of Finnish Species (Hyvärinen et al. 2019). Females generally produce only one calf per year, which makes this species particularly vulnerable to any additional source of mortality.

Today, WFR occur only in Finland, and the northwestern parts of Russia, although they previously had a wider distribution. There are currently two distinct subpopulations of WFR in Finland: Kainuu and Suomenselkä. Two decades ago, the size of the Kainuu subpopulation decreased dramatically in just a few years, while the Suomenselkä subpopulation, reintroduced 40 years ago, has increased to approximately 2 000 individuals (Paasivaara et al. 2021). Based on census data from late winter 2023, the current total number of Finnish WFR is about 3 000 individuals (Natural Resources Institute Finland, unpublished data), divided over the two main subpopulations (Fig. 1). In contrast, the Russian WFR population peaked during the 1980s. Since then, the population has decreased from 7 000 to 2 300 individuals (Panchenko et al. 2017; Danilov et al. 2020). Taking both the Finnish and Russian WFR populations into account, the total global population of WFR is approximately 5 300 individuals.

Although the main reasons for WFR population decline in Kainuu and Russia are most likely due to anthropogenic landscape change, increased predation pressure by large carnivores (especially wolves *Canis lupus*, see Kojola et al. 2004, 2009), and to a certain extent poaching (Efimov and Mamontov 2014), other factors such as traffic related mortality have affected and continue to affect the population persistence of WFR. So far, our knowledge about the survival or mortality patterns of WFR is limited. Pöllänen et al. (2023) showed that the primary cause of mortality of adult GPS-collared WFR females is predation. Accidents and traffic mortality were the second and third most important causes of deaths, and the annual mortality rate from traffic was 0.016 for GPS-collared females in both subpopulations.

As traffic mortality is practically the only direct mortality factor in WFR which can be mitigated, it is important to better understand the magnitude of this problem.

The purpose of our study was to provide basic information about the traffic mortality of WFR in Finland. Specifically, our study aimed to answer the following questions:

1. Is road traffic mortality linked to sex or age class?
2. What percentage of individuals struck in road traffic collisions died?
3. How many road traffic collisions have occurred in relation to subpopulation sizes?
4. What proportion of the two subpopulations have died because of road traffic collisions, and are there differences between the subpopulations?

In addition to road traffic mortality, we calculated the proportion of subpopulations that have died due to railway traffic collisions.

Materials and methods

Study area

We conducted our study in Finland, where two distinct subpopulations of WFR currently exist (Fig. 1, Paasivaara et al. 2021). In addition to these subpopulations, two new subpopulations are being established in western Finland by the ongoing WildForestReindeerLIFE re-introduction project (Metsähallitus, Wildlife Service Finland 2023). These new subpopulations were excluded from this study as there has been no known traffic mortality to date.

The distribution of the Kainuu subpopulation is ca. 15 000 km^2 (in Finland). During winter, WFR gather at lichen eskers found especially in the western parts of this area. In summertime, females in particular are more solitary and widely distributed across the landscape (Natural Resources Institute Finland, unpublished data). The distribution of the Suomenselkä subpopulation is ca. 40 000 km^2. During the last decade, the most important winter pastures have been near Lake Lappajärvi, situated in the south-western part of the distribution area. The summer core area of this subpopulation is in central Finland, but WFR have spread to the northeast towards the reindeer (i.e., domesticated *Rangifer*) herding area.

The density of humans is considerably lower in the Kainuu subpopulation, compared to the Suomenselkä area: 3.1 inhabitants vs. 9.1–14.3 inhabitants per km^2, depending on the province (Statistics Finland 2023). Similarly, the paved road density is lower in the area of the Kainuu subpopulation (0.14 km per km^2) than in the area of the Suomenselkä subpopulation (0.21 km per km^2) (calculated from the data presented in Fig. 2). The railway intersects both areas (Fig. 2).

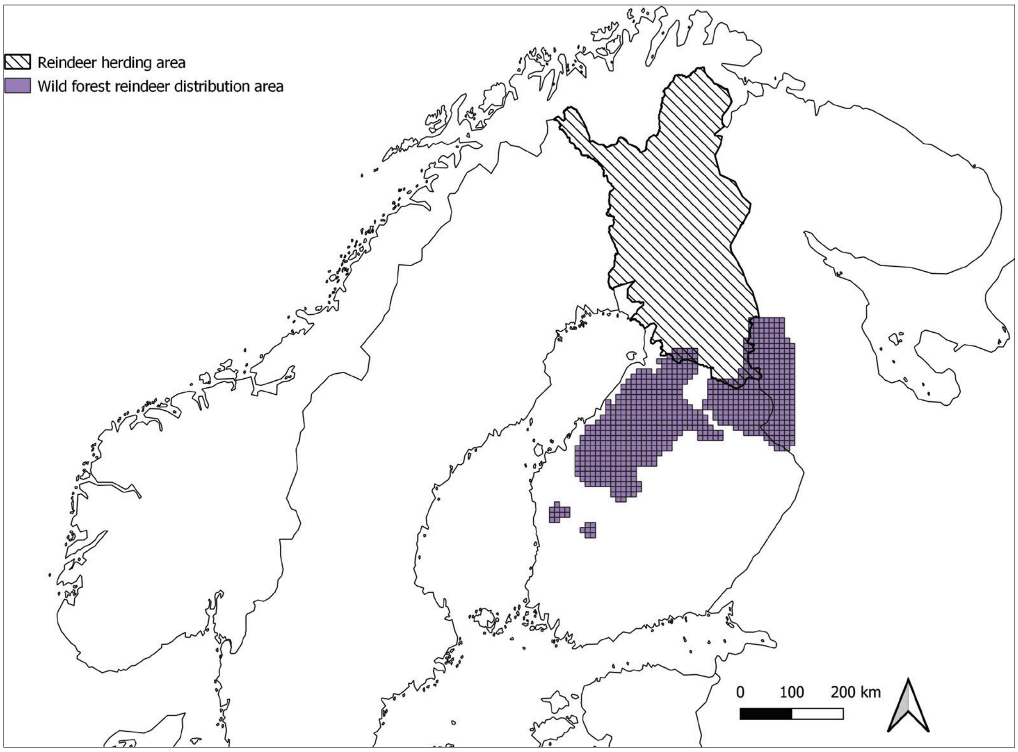

Figure 1. Subpopulations (western Suomenselkä, eastern Kainuu) of wild forest reindeer and reindeer herding area in Finland. Information is provided by Natural Resources Institute Finland (WFR distribution), Reindeer Herders' Association (reindeer herding area) and Eurostat (country borders).

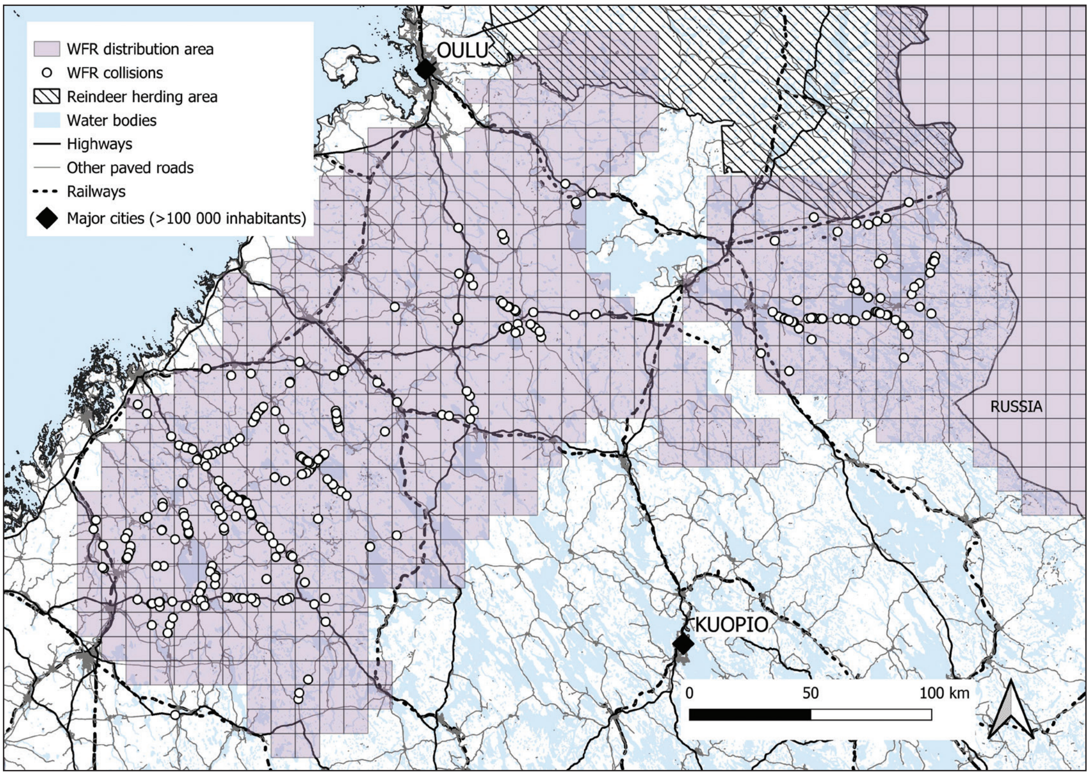

Figure 2. Paved roads, railways and wild forest reindeer -vehicle collisions within the wild forest reindeer distribution area in Finland. Information is provided by the National Land Survey of Finland (roads, railways, waterbodies, cities), Ministry of Agriculture and Forestry of Finland (WFR collisions), Natural Resources Institute Finland (WFR distribution area), Reindeer Herders' Association (reindeer herding area) and Flanders Marine Institute (seas).

Collision data

Drivers are obliged to report all ungulate-vehicle collisions to the emergency number in Finland. This information is forwarded to the Police, who in turn direct trained volunteer local hunters to provide executive assistance. Volunteers check the condition of the struck animal and euthanized it if needed. If the animal has escaped from the collision site, the volunteers attempt to track it to determine if it is injured. In addition to road casualties, volunteers are often asked to remove ungulate carcasses from the railway after collision with a train.

Since the beginning of 2017, volunteers have reported every event they have participated in, by using a mobile or computer application. Volunteers record the species, time, coordinates, and the result of the event (animal was found dead, euthanized, disappeared, not injured). The data collection is coordinated by the Game Management Associations and the database is administered by the Ministry of Agriculture and Forestry of Finland.

Collision data for 2017–2022 was extracted from the register in April 2023. First, we downloaded all collisions that occurred on a road or railway network where the species was identified as WFR. Then, we manually checked the data and excluded events (ten) which were located outside of the current distribution area of WFR (Fig. 1) because it seemed clear that the species was not registered correctly.

Population data

During the last two decades, the wintering populations of WFR have been censused 16 times in Kainuu and eight times in Suomenselkä. Depending on snow conditions, aerial censuses are conducted in late February or early March by the Natural Resources Institute Finland. Censuses are made as a total count, where the aim is to find all individuals (Paasivaara et al. 2021). In 2023, there were approximately 885 WFR in Kainuu (Natural Resources Institute Finland 2023). The latest census of the Suomenselkä subpopulation was made in 2022, when there were almost 2 000 individuals (Table 2; Natural Resources Institute Finland 2023).

For the years when aerial censuses were not conducted (Kainuu in 2018 and 2020; Suomenselkä in 2017, 2019, 2020), the Natural Resources Institute Finland (2023) has provided WFR population size estimates interpolated from the neighboring years' census results.

Data analyses

First, we calculated the percentages of adult individuals and calves in the road collision data and checked whether the sexed individuals were females or males. Then, we calculated a kill rate, a collision rate, and a road mortality rate (see Niemi et al. 2015) for both subpopulations of WFR. The kill rate was calculated from the animals struck, i.e., the percentage of struck animals that died immediately due to the collision or were euthanized afterwards. The collision rate was calculated in relation to population estimates, i.e., how many collisions occur for each 100 individuals assessed in aerial censuses or estimated wintering population (see Table 1). Similarly, the road mortality rate (road-killed individuals for each 100 individuals)

Table 1. The results of the aerial censuses of wild forest reindeer in the Kainuu and Suomenselkä areas, and annual wintering population estimates for the years when aerial censuses were not conducted (*). The information is provided by the Natural Resources Institute Finland (2023).

Subpopulation	2017	2018	2019	2020	2021	2022
Kainuu	749	732*	714	757*	799	829
Suomenselkä	1364*	1431	1610*	1789*	1968	1957

Table 2. Road-killed and struck but escaped WFR in Finland between 2017 and 2022 (total of six years), divided by subpopulations. "Condition unknown" contains individuals that escaped from the collision site and were not found by tracking (121 individuals), where the collision site was not found at all (nine cases) or information was lacking (three cases).

	Road-killed individuals (% of total)	Uninjured individuals	Condition unknown	Total number of individuals struck (% of total)
Suomenselkä	201	11	109	321 (79%)
Kainuu	58	4	24	86 (21%)
In total	259 (64%)	15 (4%)	133 (33%)	407 (100%)

was calculated in relation to aerial census data. Thus, the collision rate was always equal to or higher than the road mortality rate. For the train collision data, we calculated only a railway mortality rate. To simplify the text, we then converted the results to percentages (e.g., a traffic mortality rate of 0.1 = 10%).

We used Fisher's exact test (e.g., Ranta et al. 1999) to test possible differences between subpopulations in collision and mortality rates. Analyses were conducted using R software, version 3.1.3 (R Development Core Team 2015).

Results

A total of 407 WFR were registered as involved in 390 road traffic collisions during the six-year study period (Fig. 2, Table 2). An age class (adult/juvenile) was determined for 265 animals (65%) and the sex (female/male) for 242 animals (59%). There were 100 adult females (25%) and 110 adult males (27%) in the data, respectively. The number of calves was considerably lower; 12 female calves (3%) and 11 male calves (3%) were involved in collisions. In addition, the sex of four calves was not registered.

Altogether, 259 individuals were killed directly in the collisions or euthanized afterwards after tracking (collectively later referred to as road-killed), which yields a kill rate of 64% (Table 2). The condition of 15 individuals (4%) was checked and registered as injured. The rest of the animals, 133 individuals (33%), escaped from the collision sites and/or their condition was unknown.

In relation to wintering population sizes, the Suomenselkä subpopulation had a higher collision rate (3.0%; 3.0 collisions/100 individuals) than the Kainuu subpopulation (1.8%). The difference between subpopulations was statistically significant (DF = 1, $p < 0.001$). Concurrently, the road mortality rate was higher in the Suomenselkä subpopulation (2.0%) than in the Kainuu subpopulation (1.3%). A statistically significant difference was observed between subpopulations (DF = 1, $p = 0.003$).

WFR-train collisions occurred in both subpopulations. The total number of registered collisions was 19 in Kainuu and eight in Suomenselkä. At least 30 individuals died in these collisions in Kainuu, giving a railway mortality rate of 0.7%. In Suomenselkä, 22 individuals were reported to die in train collisions (a railway mortality rate of 0.2%).

Discussion

In this study, we gathered basic information about traffic mortality of WFR. When looking at the demographical status of road-killed individuals, the percentage of road-killed calves (those classified as juveniles) was less than five percent. This percentage was lower than the percentage of calves in the population; for example, in the aerial census conducted in April 2021, the percentage of calves was 14.0% in Kainuu and 13.5% in the Suomenselkä subpopulation (Paasivaara et al. 2021). Our dataset was too small to draw any firm conclusions, but the results suggest that the road mortality of WFR might be adult biased. If this is correct, the possible effect of road mortality on population persistence could be larger than the collision numbers indicate. Adult females in particular are critical to ungulate population growth and survival (Gaillard et al. 1998).

Traffic mortality, unlike predation-related mortality, does not appear to be linked to an animal's physical condition, at least in some circumstances. Gunson et al. (2022) found that road-killed elk were in better physical condition compared to individuals killed by predators, and therefore suggested that vehicle collisions are an additive source of mortality. Although our data did not detail the physical condition or accurate age structure of road-killed WFR, a study by Karhula (2021) found that adult road-killed WFR were younger than individuals killed by predators. This indicates that the expected contribution of road-killed WFR to the population growth will be missed.

Our finding that there were almost an equal number of adult females and males in the collision data was somewhat surprising. There are slightly more females than males in the WFR population (Natural Resources Institute Finland, unpublished data), but the ratio of close to 1:1 was still unexpected. Ungulate males are often over-represented in collision statistics in relation to the demographic population structure (Etter et al. 2002; Lopez et al. 2003; Olson et al. 2014; Gunson et al. 2022; but see Madsen et al. 2002), which is probably at least partly due to their longer daily movements, especially during the rut (Webb et al. 2010; Niemi et al. 2013). One possible explanation for our results could be the strong herding behavior in WFR, especially in winter, which could lead to both sexes crossing roads in equal numbers. Our observation underlines again the relative importance of traffic mortality for the WFR population; it seems that the most valuable individuals for the population (adult females) are also relatively vulnerable to traffic mortality.

We found a kill-rate of 64%, i.e., approximately six out of ten WFR struck died due to road traffic collisions. This is a much lower rate than reported in earlier UVC-studies; for example, Almkvist et al. (1980) found a kill-rate of 94% for roe deer and > 80% for moose *Alces alces*. However, only 4% of struck WFR were classified as uninjured in our data, which means that approximately one third of hit individuals disappeared after the collision. It is not known what proportion of these animals were fatally wounded and would have died later. It could be speculated that the percentage might be high; WFR tend to move and cross roads as a herd especially in wintertime, and tracking an injured individual among others can be impossible. In addition, not all accidents are reported to start with (Bíl and Andrášik 2020). We therefore note that even though the kill-rate we found was relatively low, the true number of individuals which are killed in road traffic is probably more or less the same as the number of collisions. Thus, if we want to evaluate the proportion of road killed individuals in WFR populations, reporting a collision rate might be a better indicator.

When looking at the subpopulation level, the collision rate in relation to the wintering population size was higher in the Suomenselkä subpopulation than in Kainuu (3.0% vs. 1.8%). This was true also for road mortality rates (2.0% vs. 1.3%). The observed proportions were approximately the same as those reported elsewhere for other ungulate species. Seiler et al. (2004) estimated that 4% of the Swedish moose population was killed in road traffic. In a study conducted in a densely populated area in southern Finland, Niemi et al. (2015) reported road traffic mortality rates ranging from 2.1% to 6.5% of the wintering population, depending on the ungulate species. The most likely factor explaining the observed differences between WFR subpopulations is traffic volume, which is known to affect the number of ungulate-vehicle collisions (Seiler 2004; Bíl et al. 2021). There are more major roads in the distribution area of the Suomenselkä subpopulation than in Kainuu, and WFR in Suomenselkä are therefore more likely to cross roads during their daily routines and seasonal movements.

Even though our study was based on only six years of data, the high proportion of adult road-killed WFR implies that traffic mortality should be seriously considered as a conservation issue. This calls for species-specific mitigation measures for WFR, which differs from other wild ungulates as a strongly migratory, herding species. While widely used methods such as under and overpasses with wildlife fences (Clevenger 2005; Olsson and Widen 2008; Huijser et al. 2016) are not cost-effective mitigation measures for WFR moving long distances and crossing some (non-fenced minor) roads maybe only twice per year (but see Sawyer et al. 2012, 2016), identifying high risk road sections might be a key to target other measures such as wildlife detection systems and short-term temporal warning signs (Huijser et al. 2015). In the future, intelligent systems incorporating sensor technologies and machine learning (reviewed by Nandutu et al. 2022) will be increasingly used to detect road-crossing animals, including WFR, to give drivers more time to react.

In this study, we concentrated mainly on road-traffic mortality of WFR because the volunteer-based UVC-data collection system used in Finland is inadequate for collecting a comprehensive dataset from railroads. However, as our limited data showed, WFR are occasionally also killed in train collisions, sometimes several individuals at one time. Rolandsen et al. (2015) highlight the importance of similar factors affecting both road and railway collisions, such as animal population density, and traffic intensity. For semi-domesticated reindeer, they found a positive correlation between the frequency of collisions and reindeer density, but only in areas where the railway crossed the winter range. Understanding the factors that contribute towards WFR railway mortality will allow us to develop mitigation measures for railway systems as well. As with the road network, the first step will be to recognize collision hotspots and then apply mitigation measures such as thermal cameras and early acoustic warning (Bhardwaj et al. 2022) to reduce the risk of WFR-train collisions.

Our study provides only the very first information about traffic mortality of WFR. The dataset we used should be better utilized in the future, for example to recognize spatial collision hotspots (Shilling and Waetjen 2015; Bíl et al. 2019), and again highlight the areas with high risk of WFR collisions (e.g., Morelle et al. 2013). By combining the collision data with different landscape and forest structure datasets, we could study how landscape and habitat variables affect the location of collisions (Danks and Porter 2010; Galinskaitė et al. 2022) and try to predict future collision hotspots based on the environmental and road-related variables (Laube et al. 2023). Also, the existing collision data provides an opportunity to study seasonal patterns of WFR collisions in a more detailed way.

We were only able to use WFR collision data in this study. In the future, it would be useful to use collision data together with GPS data from satellite-collared WFR. This would help us identify areas where animals are more likely to cross the main roads during their annual movements. It is noteworthy that the potential crossing sites or areas are not necessarily the same as indicated by the animal-vehicle collision data (Neumann et al. 2012; Lee et al. 2023), which shows that collision data alone does not provide us with a complete understanding. In addition, GPS-data would help us to recognize the effects of roads and traffic on individual movements (see Wilson et al. 2016), to better understand how anthropogenic factors affect the behavior and survival of WFR.

Conclusions

Here, we studied traffic mortality of wild forest reindeer in Finland by using collision data which was collected during 2017–2022 by volunteer hunters. Our results indicate an adult bias in road mortality. Interestingly, adult females and males were almost equally represented in the collision data, unlike other ungulate traffic mortality studies where males are often over-represented. We have suggested that this may be due to the strong herding behavior exhibited in WFR, especially in winter. Although our dataset was limited, these results cumulatively suggest the relative importance of traffic mortality for WFR population persistence. In particular, the most valuable individuals for the population, adult females, appear to be relatively vulnerable to traffic mortality. Future studies may further use this dataset to focus on seasonal patterns of WFR collisions in a more detailed manner and predict collision hotspots. Such studies could help to plan and locate species-specific mitigation measures to reduce traffic mortality of this endemic ungulate species.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

Milla Niemi conceived the study, performed the data analysis, took the lead in writing the manuscript with support from Sari C. Cunningham and Robert Serrouya, and designed the figures. All authors read and revised the manuscript, and approved the submitted version.

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Data availability

All of the data that support the findings of this study are available in the main text.

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Research Article

Biodiversity monitoring with intelligent sensors: An integrated pipeline for mitigating animal-vehicle collisions

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Abstract

Transports of people and goods contribute to the ongoing 6th mass extinction of species. They impact species viability by reducing the availability of suitable habitat, by limiting connectivity between suitable patches, and by increasing direct mortality due to collisions with vehicles. Not only does it represent a threat for some species conservation capabilities, but animal vehicle collisions (AVC) is also a threat for human safety and security in transport and has a massive cost for transport infrastructure (TI) managers and users. Using the opportunities offered by the increasing number of sensors embedded into TI and the development of their digital twins, we developed a framework aiming at managing AVC by mapping the collision risk between trains and ungulates (roe deer and wild boar) thanks to the deployment of a camera trap network. The proposed framework uses population dynamic simulations to identify collision hotspots and assist with the design of sensors deployment. Once sensors are deployed, the data collected, here photos, are processed through deep learning to detect and identify species at the TI vicinity. Then, the processed data are fed to an abundance model able to map species relative abundance around the TI as a proxy of the collision risk. We implement the framework on an actual section of railway in south-western France benefiting from a mitigation and monitoring strategy. The implementation thus highlighted the technical and fundamental requirements to effectively mainstream biodiversity concerns in the TI digital twins. This would contribute to the AVC management in autonomous vehicles thanks to connected TI.

Key words: Abundance modelling, animal vehicle collision, autonomous vehicle, camera traps, computer vision, connected transport infrastructure, deep learning, digital twin, risk management, ungulates

Introduction

Transports of people and goods contribute to the ongoing 6th mass extinction of species (Forman and Alexander 1998; Holderegger and Di Giulio 2010; Haddad et al. 2015, IPBES 2019; Grilo et al. 2021). They impact species viability

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by three main processes (Teixeira et al. 2020). Transport infrastructure (TI) can have an impact on species: 1) by reducing the availability of suitable habitat for species (Ouédraogo et al. 2020; Kroeger et al. 2021; Fischer et al. 2022; Remon et al. 2022), 2) by limiting the functional connectivity between patches of suitable habitat (Ujvári et al. 2004; Balkenhol and Waits 2009; Safner et al. 2011; Remon et al. 2018, 2022), and 3) by increasing direct animal mortality due to collisions with vehicles (Ceia-Hasse et al. 2018; Testud and Miaud 2018; Lehtonen et al. 2021; Moore et al. 2023).

Not only do they represent a threat for some species conservation capabilities, animal vehicle collisions (AVC) are also a threat for human safety and security in transport when large species are involved. Animal vehicle collisions' events also represent a massive cost for TI managers and users due to infrastructure and vehicle repair or compensations for damages (Huijser et al. 2009). For instance, bird strikes represent a 1.2 billion US$ cost annually to the aerial transport sector (Allan 2000) and caused more than 700 human deaths since 1905 (Avisure 2019; Metz et al. 2020). Moose road-kills along a 61 km railway in central Norway cost 250 000 US$ annually (Jaren et al. 1991).

In Europe, terrestrial AVC often involve large mammals (Grilo et al. 2021) such as moose (*Alces alces*), roe deer (*Capreolus capreolus*), or wild boar (*Sus scrofa*). Animal vehicle collisions also impede conservation programs across the EU, particularly concerning large carnivores like grey wolf (*Canis lupus*), brown bear (*Ursus arctos*), or Eurasian lynx (*Lynx lynx*) (Bauduin et al. 2021; Grilo et al. 2021). In addition, large mammal populations tend to increase across the EU. For instance, Ledger et al. (2022) highlighted respectively a 331% and 287% increase of the red deer and roe deer population in the EU. Thus, a solution to ensure traffic safety without enclosing the transport network should be found to limit the barrier effect of transport infrastructure on large mammals without increasing, and rather ultimately reducing, the number of AVC (Grilo et al. 2021; Seiler et al. 2022).

The transport system is in a deep digital transformation with the development and deployment of data-driven TI management (ITF 2021). Thus, an increasing number and diversity of sensors is embedded into TI providing time-continuous information to TI managers ultimately through the TI's digital twin (DT) which is the digital representation of the physical TI (Grieves 2016; Batty 2018; Singh et al. 2021). Indeed, future roads are expected to become able to produce their own energy, be self-monitored thanks to multiple embedded sensors, be carbon neutral and ensure biodiversity gain. Such an autonomous system is expected to also produce multiple services thanks to its digital copy collecting and analysing the sensors' data (Hautière et al. 2012, 2023, ITF 2023). To date, collected data are mainly used for TI maintenance or user safety (Moulherat et al. 2022). In addition to the TI management, connected TI are expected to provide information to the vehicle which, in turn, would become more and more autonomous in the near future (Seiler et al. 2022, ITF 2023). In this perspective, sensors embedded in the TI are providing the infrastructure digital model with data collected and analysed for providing relevant information that can feed the TI users including vehicles and therefore drivers (ITF 2021, 2023).

Unfortunately, biodiversity concerns are not yet part of this TI digital environment which nevertheless offers a suitable place for biodiversity-based risk management such as AVC (van Eldik et al. 2020; ITF 2021, 2023; Djema

2022; Moulherat et al. 2022). Indeed, sensor-based animal recognition ability, thanks to artificial intelligence and, particularly, deep learning, is growing very fast (Tuia et al. 2022) making it possible to automatically detect and recognise the main species involved in AVC in the EU (Aodha et al. 2018; Demertzis et al. 2018; Rigoudy et al. 2022). From a TI management perspective aiming at reducing AVCs, the main current applications are, to date, based only on large mammal detection and aimed at informing drivers of the presence of a big animal. The animal detection can be used to animate dynamic panels or to threaten individuals approaching the TI with a combination of light and sounds with sometimes limited efficiency (Seiler and Olsson 2017). Collecting and analysing species detections (and non-detections) provided by sensors in the TI DT would contribute to improve the AVC management. Indeed, once identified, a collision risk map may be produced by models able to approximate the passage rate of the species involved in AVC around the TI. In this perspective, occupancy or abundance modelling can produce spatial estimates of presence probability or abundance, respectively (Burton et al. 2015; Gilbert et al. 2020; Gimenez et al. 2022; Tuia et al. 2022). Such maps would therefore provide drivers and connected vehicles with relevant context information about the actual risk of species involved in AVC presence.

With the OCAPI initiative, the goal is to enhance the integration of biodiversity-oriented digital facilities into the DT of TI (Moulherat et al. 2021). In this paper, we develop a framework aiming to provide large mammal's presence risk in the TI vicinity based on sensor-based monitoring system. The framework is applied on an actual AVC hotspot between ungulates (roe deer and wild boar) and trains in south-western France benefiting from a long-term mitigation measures program (see Suppl. material 1 for further information about the long-term program). In this context and based on the monitoring program planned as well as simulation of spatially explicit ungulate's population dynamics implemented in 2021, we simulated ungulates detection stories, mapped their presence risk close to the TI, and tested the model performances to predict the theoretical AVC risk. Then, in 2022–2023, after monitoring for a single year, we applied the theoretical framework to the real situation to test the system for further improvements.

Methods

The methodological framework developed and implemented in this study is composed of 5 major steps (Fig. 1). This framework begins with a sensor-based monitoring design phase (step 1 to 3) based on population dynamic simulations of focal species (step 1). The framework then tests the monitoring design expected efficiency in an iterative process (steps 2 and 3). Step 4 of the framework is dedicated to sensor-based data processing, thanks to deep learning, which, in turn, feed abundance models, providing a proxy of the AVC risk (step 5).

Study site

The study site is a 19.7 km section of the railway joining Toulouse to Agen in south-western France (Fig. 2). This section supports about 24 trains daily and was identified by the TI managers for its frequent collisions with large

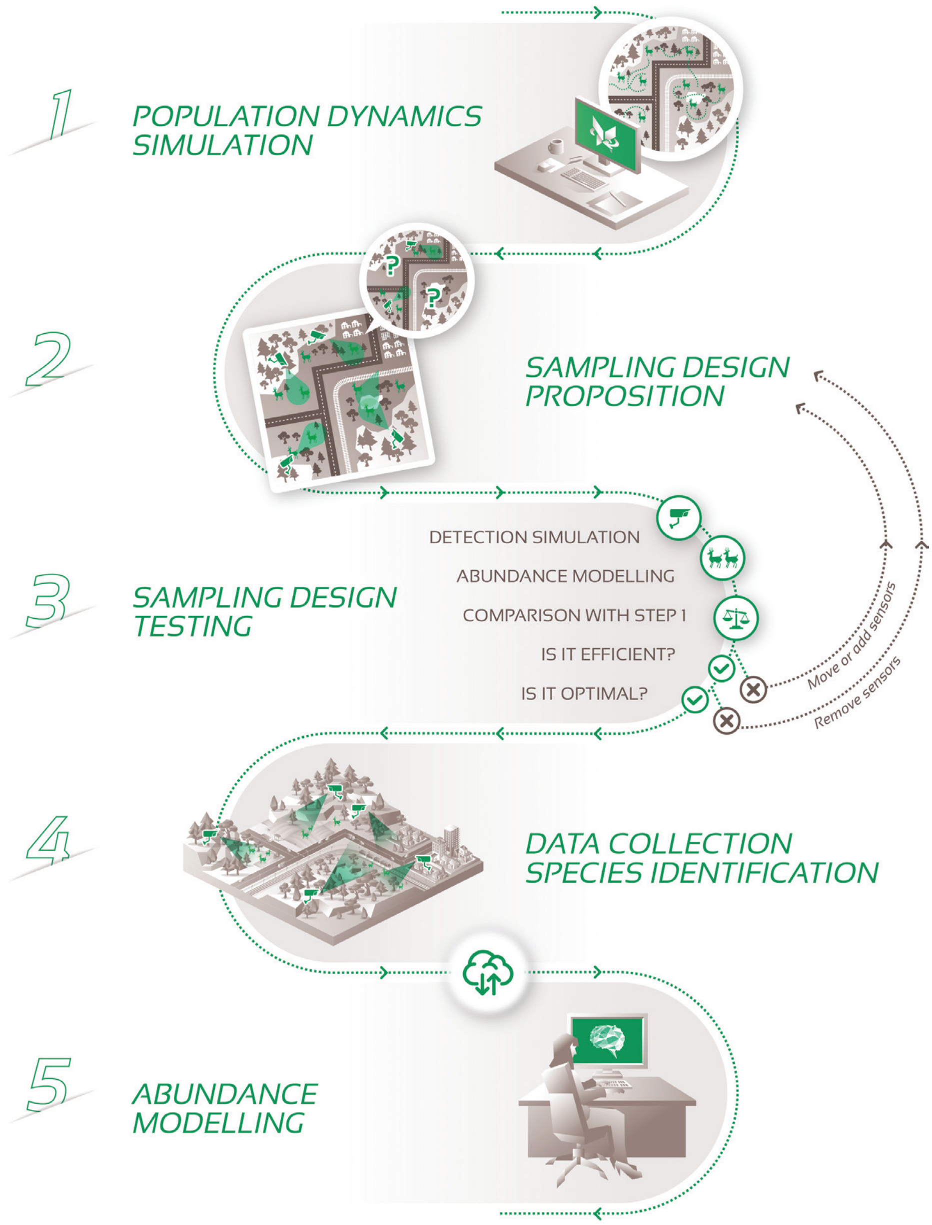

Figure 1. Framework to deploy sensors along a transport infrastructure to map the animal abundance in the transport infrastructure vicinity in order to manage the animal vehicle collision risk.

wild mammals (mainly roe deer and wild boar). This site is part of a regional AVC reduction program launched by the French railway network management company (SNCF Réseau) in 2018 (see Suppl. material 1 for details about the comprehensive program). The program concerns 44 strategic sites with a high number of AVC, where a statistical analysis of collisions' conditions has been performed (Gaillard 2013; Saint-Andrieux et al. 2020) and combined with spatially explicit population dynamic simulations of ungulates to identify the most sensitive places to AVC (Boreau de Roincé et al. 2018). For 5 of them, scenarios of mitigation measures have been proposed and their cost-efficiency evaluated based on the expected population functioning after scenarios implementation thanks to new simulations (Zurell et al. 2021; Moulherat et al. 2023). At the same

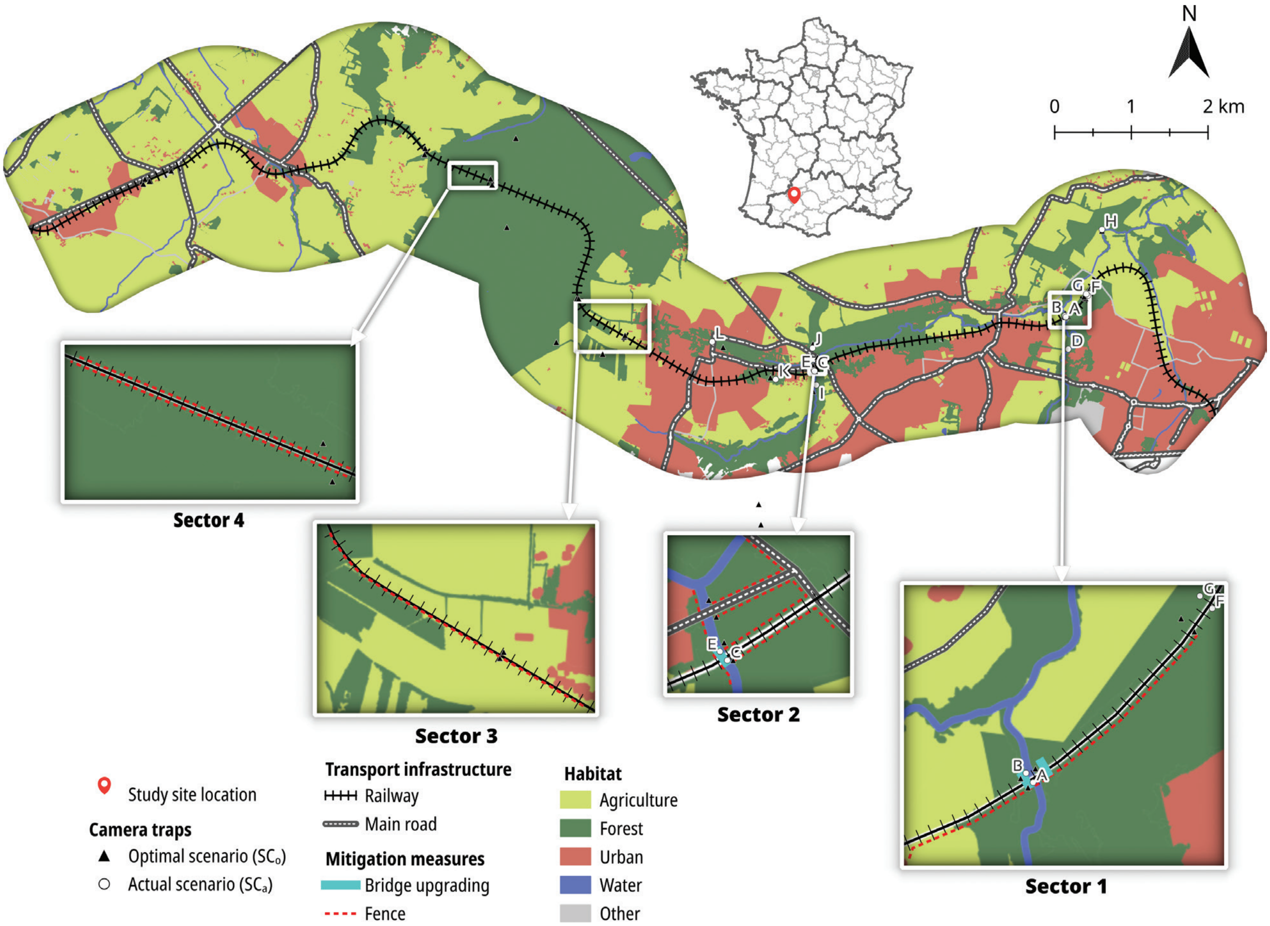

Figure 2. Study site in south-western France focused on 19.7 km of railway where numerous AVC occurred the last 10 years. The land cover is represented using the 5 main habitat typologies as used for the statistical analysis. Camera traps deployed on the field are identified by a letter from A to L.

time, a regional camera trap monitoring program following a Before After Control Impact (BACI) design (Smith 2002) was designed to evaluate the mitigation measures efficiency. The main mitigation measures planed on the study site are the upgrading of two existing bridges by reshaping the bridges' embankment (sectors 1 and 2, Fig. 2) and the fencing of 4 sections of the railway to drive animals to existing or upgraded passages or safer crossing places (sectors 1, 2, 3 and 4, Fig. 2). The work concerning the bridges upgrading is planned for 2025.

The study site benefits from a land use map produced by combining data from Corine Land Cover (Büttner et al. 2017), BD TOPO® (IGN 2021), ROUTE 500® (IGN 2020), dedicated fieldwork, and photointerpretation within a 5-km buffer zone around the 19.7 km of the studied railway section. Habitats have been characterised into 26 classes based on the standard EUNIS typology.

Ungulate population dynamic simulation

As a part of the AVC hotspot identification, we used SimOïko to perform spatially explicit population dynamic simulation of ungulates on the study site. SimOïko is an individual-based spatially explicit model developed to perform population viability analysis based on the MetaConnect model

(Moulherat, 2014). In the model, each individual of the simulated population is a unique agent whose virtual life is driven by stochastic processes. For example, survival of an individual depends on the result of a Bernoulli event with probability *p* corresponding to the average survival of the individual age class. The model assumes that individuals live in panmictic patches of suitable habitat. In this study, roe deer and wild boar, the AVC target species, are not explicitly modelled. Instead for the sake of simplicity, we used a virtual species representative of a mixture of roe deer and wild boar life history traits (Caro and O'Doherty 1999; Caro et al. 2005; Baguette et al. 2013) hereafter called ungulate. Suitable patches for ungulate in this landscape are expected to be forests and shrublands.

We modelled the dispersal behaviour of ungulate moving between suitable habitat patches using the SimOïko embedded Stochastic Movement Simulator (SMS) algorithm (Palmer et al. 2011). The SMS algorithm assumes that individuals can perceive their environment to a certain distance and tend to use the "easiest" path within this perceptual range. In this respect, the model needs a rugosity map reflecting the ability of individuals to cross the different types of land cover existing within the study site landscape matrix. Thus, for each of the 26 natural habitat types of the study site, a rugosity coefficient is assigned based on expert opinion on ungulate moving abilities (Dutta et al. 2022) (see Suppl. material 1 for the comprehensive parameterisation of SimOïko). SimOïko's input maps are rasterized using a 5×5 m pixel resolution.

Simulations were initialised with 118 individuals assuming that all the potential suitable patches are occupied at their maximum carrying capacity. The simulation runs for 100 years which is sufficient to ensure the metapopulation dynamic stabilisation for at least the last 50 years (see Suppl. material 1). Therefore, only the results from the last 50 years were used. Simulations were repeated 50 times.

As a result, the model provides the expected number of individuals living in the studied landscape and a map of the cumulative number of animal passage per map pixel during the simulation time (Moulherat 2014).

Monitoring strategy

To map the abundance of ungulate in the TI vicinity using the camera traps deployed for another purpose (e.g. evaluate the mitigation measures efficiency), we mimic the expected monitoring process and analysis to evaluate its effectiveness in an iterative four-step process:

1. Propose a location of camera traps scenario.
2. Use the camera trap location scenario and the movement simulation results to simulate detection stories.
3. Analyze the simulated monitoring results with abundance modelling.
4. Compare the movement simulation and the abundance model results in order to control the monitoring program ability to be used for mapping the abundance of ungulates. If not, come back to step 1 if some adaptations are possible, otherwise the ability to actually map the ungulate's abundance is not expected.

Monitoring program

On the study site, we designed a monitoring program to evaluate the efficiency of 2 bridges upgrades (including fencing) (sectors 1 and 2 Fig. 2) and the fencing only of 2 additional sections (sectors 3 and 4 Fig. 2) in reducing AVC. Each section benefiting from a mitigation measure is expected to be monitored by a network of minimum 6 cameras. A couple of cameras are recording each side of the railway (entrances of bridges or observed animal's tracks on the field for the fencing projects) to monitor crossing events. Two other cameras are deployed in forests, between 177 and 651 m from the railway, as controls of the ungulate activity in the surrounding suitable habitats (Fig. 2). Another pair of cameras are placed to survey crossing events in sections not benefiting from mitigation measures as a control of the crossing activity. Additional cameras are added to monitor crossing events in sections not benefiting from mitigation measures, but with suspected high crossing frequency or for which simulations' results show a possible crossing location deferment. Thus, the total program comprises 38 cameras each deployed for 5 years minimum and hereafter called Optimal scenario (SC_o).

The monitoring began in August 2022. However, due to TI manager investment abilities, the monitoring could only start for the two bridges upgrading reducing the study site section to 11.7 km- long for the framework showcasing (sectors 1 and 2 Fig. 2). The continuous deployment of 12 camera traps (Bolyguard, MG984G-36MP 4G) required to monitor these two sections, will be maintained for at least 5 years by the local hunter association and is defined as the actual scenario (SC_a).

Both scenarios of camera trap deployment (SC_o and SC_a) were evaluated for their expected ability to provide relevant mapping of ungulate abundance close to the TI.

Virtual and actual camera-trap data processing

Frequentation story simulation of the virtual camera traps

We used the simulated frequentation map to mimic a camera trap survey leading to a frequentation history of 30 recording occasions. Thus, for each sampling occasion, the number of detections in a pixel containing a camera trap is simulated as a random event following a Poisson distribution. The average value of this distribution corresponds to the average number of passages of ungulates within the pixel during a single time-step of the population dynamics simulation. In this respect, we divided the average number of passages of dispersing individuals by the proportion of dispersing individuals.

Deep learning algorithm training for wild boar and roe deer automatic detection

To recognise the main species (here roe deer and wild boar) involved in AVC on the images produced by the monitoring program, we used the YoloV8 deep neural network (Jocher et al. 2023). This model is known to be fast and accurate for detecting and classifying objects in images. The model finds objects of interest in a picture and creates a bounding box around them. Then the model assigns a category to the bounding box such as a species name in this work. In this perspective, we fine-tuned a YoloV8 pre-trained on the COCO data set (Jocher et al. 2023) with the project data set (Weiss et al. 2016).

The project data set is composed of 40 358 images provided by 41 data providers across France and annotated by 51 experts thanks to the project's collaborative annotation platform (www.ocapi.terroiko.fr). This data set was completed by the images of the COCO data set containing animals or vehicles. Annotations consist in bounding boxes drawn on the pictures and labelled with the name provided by the French national taxonomic referential (Gargominy et al. 2021). The dataset was split randomly into a train (80%) and validation (20%) data set. The train data set contained 262113 boxes from 26 labels including 1307 boxes of wild boar (*Sus scrofa*) and 418 boxes of roe deer (*Capreolus capreolus*). Approximately 5.5% of the images were empty (no animals, humans or vehicles). Other frequently observed labels included humans, vehicles, foxes, badgers, dogs, cats, horses, chamois, lynx and leporidae, among others. We used an independent data set as test. The test data set is composed of 1174 images containing 212 boxes of roe deer and 24 of wild boar. Thirteen other species with an average of 72.8 boxes (ranging from 1 to 188) per species are present in the test data set.

Frequentation story of the deployed camera traps

Here we used the photos taken from 29 August 2022 to 16 April 2023 (33 weeks) for 11 sites, and from 24 October 2022 to 16 April 2023 (25 weeks) for the site E to test the framework in real conditions. The local hunter association made simple annotations by identifying the species seen on the pictures (no bounding boxes) using 3 classes labelling system: ungulate (roe deer and wild boar), human/vehicle and other, including any other species and the empty pictures. The data set thus produced is then called the showcase data set. When observations were closer than three minutes apart, only the first observation was kept as the camera-trap was likely triggered several times by the same individual (Rovero and Zimmermann 2016). The observations were discretised into weekly intervals to generate the detection history, which records the number of ungulate detections per week and camera trap site.

Abundance modelling

In this paper, we do not aim to estimate the absolute ungulate abundance within the study site, but rather spatially estimate their relative abundance to identify the places with higher collision risks. To do so, we used the N-mixture model proposed by Royle (2004). In this respect, the study area was split into hexagonal cells of 200 m large, leading to 3.5 ha cell's area. The analysis was performed in R version 4.3.0 (R Core Team 2023) using the pcount function from the unmarked package (Fiske and Chandler 2011; Kellner et al. 2023).

To test the monitoring design efficiency, we compared the normalised simulated spatial pattern of ungulate movements with the normalised abundance predicted by two models using different covariates. The first model (*Mod1*) is built with a single site covariate: the sum of the movements in the cell during all time-steps of all repetitions. The number of sensors per cell is also used as detection covariate in *Mod1*. The second model (*Mod2*) is based on ecological covariate rather than population dynamic simulation output. *Mod2* used several spatial covariates extracted from the land use map:

- The percentage of agriculture, forest, urban and water in each cell.
- The distance between the camera traps and the closest agriculture, forest, railway, road, urban area, water (for model parameters' optimisation).
- The distance between the cell centroid and the closest agriculture, forest, railway, road, urban area, water (for prediction over all the map).

We performed a PCA with the areas of agriculture, forest, urban, water per cell to reduce the number of variables explaining landscape variability in the area while managing the correlation between variables (Gimenez and Barbraud 2017). The two first principal components were kept, representing, respectively, 84,4% and 12,9% of the variance. The first principal component mainly represents the gradient between forests and urban areas, whereas the second represents the gradient between agricultural areas and the other habitats. We therefore used the cell coordinates on these two axes as synthetic uncorrelated descriptors of the cell habitats' characteristics. In the spirit of principal component regression (Graham 2003), the model's covariates were selected on the basis of their predictive capacity, according to the Akaike information criterion (AIC) (Akaike 1974; Burnham et al. 2002), and their ability to represent the variability of the habitats in the study area. For abundance covariates, the distance to each habitat and the two synthetic variables were tested. For detection covariates, the average weekly temperature and the weekly rainfall were tested. We selected the model covariates based on the actual frequentation story. The final model is built of three covariates, the two synthetic covariates from the PCA and the distance to the railway. Only *Mod2*, was used to map the actual abundance of ungulates.

Results

Testing the sampling design

The simulation process aiming at mimicking the camera trap survey under the SC_o scenario is composed of 27 sites with 1 to 3 camera per site. The average detection per sampling occasion is of 21.2 occurrences (ranging from 0 to 90 occurrences).

Considering the SC_a scenario, based on 12 sites with a single camera, the average detection per sampling occasion is 11.5 occurrences (ranging from 0 to 29 occurrences). With both scenarios, all sites benefit from at least one detection.

Modelling the simulated abundance of ungulate with simulated frequentation stories

The sampling effectively catches most of the overall simulated movement patterns, both with the expected (SC_o) and actual (SC_a) sampling (Fig. 3). Both protocols identify the same potential collision hotspots due to higher ungulate abundance (Fig. 3). The *Mod1* model prediction is similar to the population

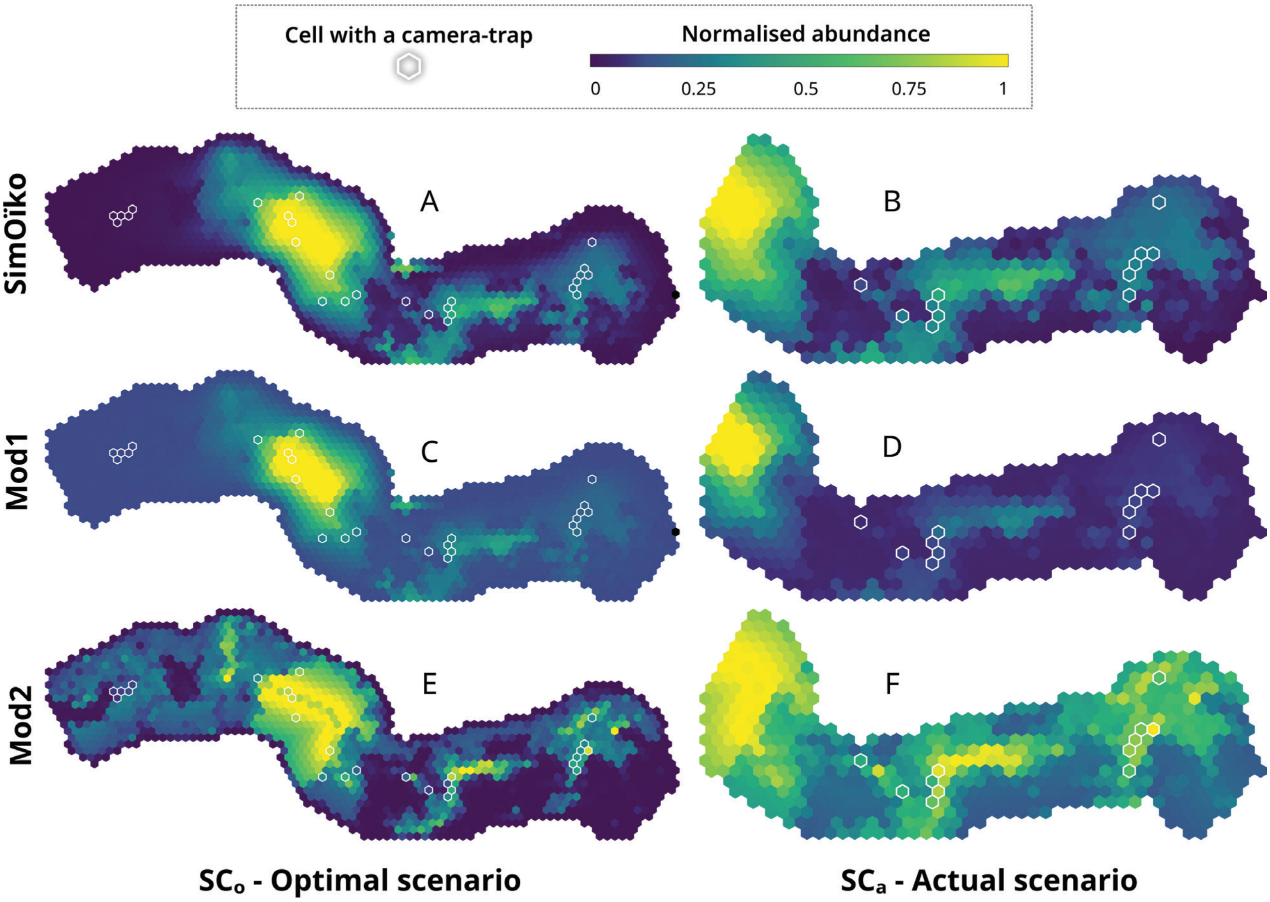

Figure 3. Normalised relative abundance of ungulates per 3.5 ha cell simulated by the population dynamic model (panels **A** and **B**), the *Mod1* abundance model (panels C and D) and the *Mod2* model (panels **E** and **F**) for SC_o (panels **A, C** and **E**) and SC_a (panels **B, D** and **F**). For comparison purposes, the normalisation was performed by normalising each cell of a map by the 97.5 percentile value. Regardless of the abundance modelling scenario, the sampling scenarios are expected to be able to identify relatively the riskiest sectors.

dynamic simulation results under SC_o. However, under SC_a, the global pattern also corresponds to the initially simulated pattern but the lack of cameras in cells mainly composed of forest habitats with very high simulated frequentation over-concentrates the abundance prediction in a limited number of cells. With *Mod2*, the global pattern leads to similar most frequented places in the landscape as *Mod1* and the population dynamic results for SC_o and SC_a. While *Mod1* over- concentrates the abundance in a limited number of cells compared to the simulation results, *Mod2* tends to retrieve a similar abundance general pattern but to over-spread the abundance around the high abundance cores.

Estimating the actual abundance of ungulates

Automatic species recognition

On the OCAPI data set, the mAP@0.5 metric (mean average precision when the intersection over union (IoU) (Padilla et al. 2020), is at least 0.5) of the classification model is 0.78 (Everingham et al. 2010). The confusion matrix is built using the default parameters from YoloV8 (confidence threshold = 0.25, IoU threshold

= 0.45). With precisions (Padilla et al. 2020) higher than 90% and recall (Padilla et al. 2020) ranging from about 80% to 97%, the model properly recognises the targeted species (roe deer and wild boar) (Table 1). Using the model on the test data set, performances to recognise roe deer and wild boar fall down, highlighting the model's lack of generalization ability (see Suppl. material 1).

Considering the showcase data set, with 80.8% of good classification when an ungulate is actually present on the pictures (Fig. 4), the model provides useful information to map the AVC risk. For 15.7% of the ungulate observation prediction, the picture is actually empty or contains another species (mainly badger confused with wild boar, see Suppl. material 1). Fig. 4 also points out the model's ability to identify humans and vehicles as well as other animals and empty pictures.

Table 1. Classification model performance. The precision reflects the model ability to limit the false positives' prediction while the recall corresponds to its capability to avoid false negatives.

	Validation data set			Test data set		
	Number of annotations	**Precision (%)**	**Recall (%)**	**Number of annotations**	**Precision (%)**	**Recall (%)**
Roe deer	93	92.47	79.63	212	74.06	83.51
Wild boar	352	93.18	90.11	24	79.17	19.39

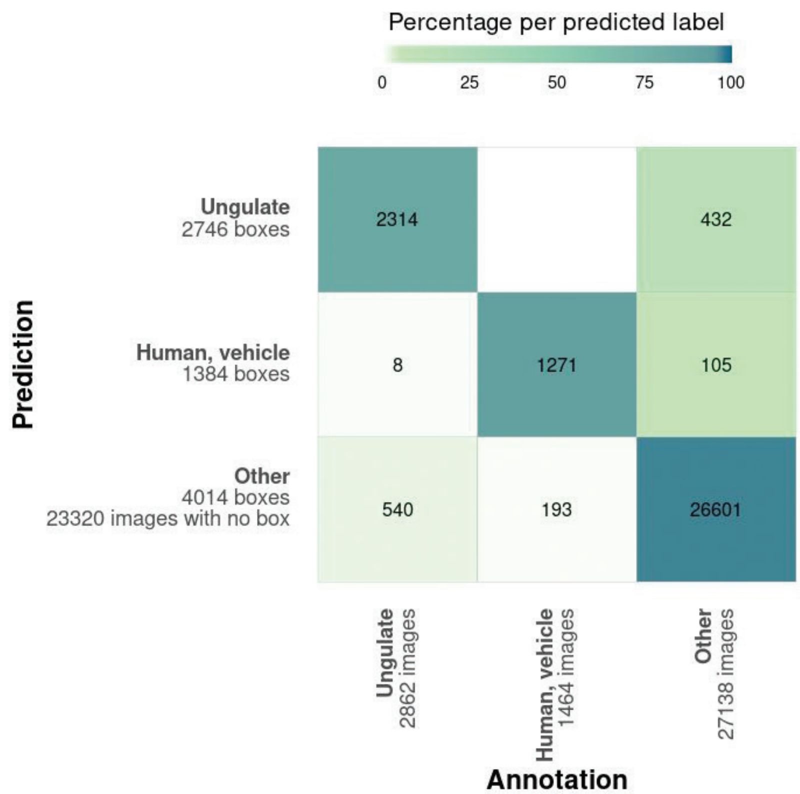

Figure 4. Comparison between prediction made by the model and the actual annotations performed by the local hunter association on the showcase data set. Pictures containing roe deer or wild boar are grouped as ungulates. Similarly, the predicted "Other" class merges boxes with other animals and empty pictures. Thus, the model predictions are presented under a form comparable to the one used by the hunter association. Details of the showcase data set processing results are developed in Suppl. material 1.

Mapping the actual abundance of ungulates

Mod2, implemented on the data issuing from the available 33 weeks monitoring program, results in ungulates concentrated along the two rivers crossed by the railway and in the Bouconne forest in the western part of the site (Fig. 5).

Discussion

In this paper, we associated methods from ecology, data science and engineering to develop a 5-steps framework for AVC management on a linear transport infrastructure (Fig. 1). Our showcase was developed on a railway section but the framework fits with any type of transport infrastructure (see Suppl. material 1). Developing and actually implementing this framework on the field demonstrates that managing the AVC risk thanks to appropriate sensor deployment and data analysis is challenging (see Suppl. material 1) but possible. However, the showcase highlights that many technical as well as fundamental improvements are required before deployment may be possible in future transport infrastructures.

Embedding biodiversity relevant sensors into the infrastructure

We implemented the framework for an existing TI benefiting from a specific monitoring program. Because biodiversity monitoring is not the central job of TI managers, we can hardly expect that they would deploy a sensor network specific for that purpose. Thus, our framework was developed to be conveniently part of an existing network dedicated to other goals (here evaluating the mitigation measures efficiency). However, steps 1 to 3 (the sensor-based monitoring design phase) may be part of the TI conception phases and particularly contribute to environmental impact assessment. Indeed, population modelling is increasingly used for decision making including an environmental impact assessment

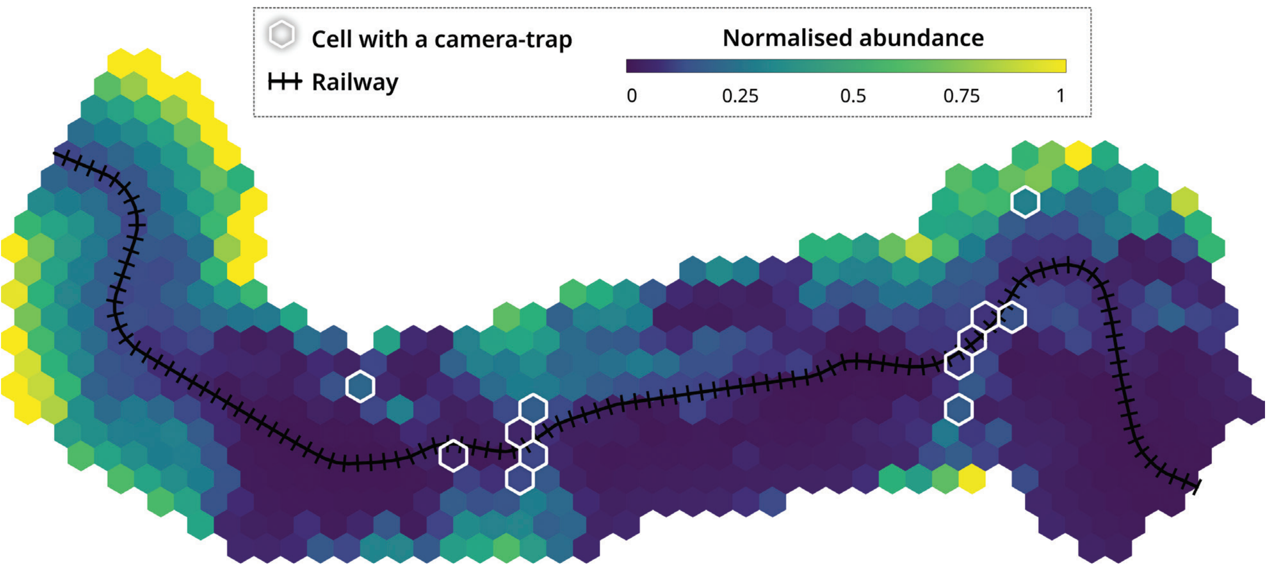

Figure 5. Normalised relative abundance of *ungulates* per 3.5 ha cell estimated by the *Mod2* model. *Ungulates* abundance is used as an AVC risk proxy along the railway section. The higher the abundance, the higher the AVC risk.

(Tarabon et al. 2021; Zurell et al. 2021; Boileau et al. 2022; Moulherat et al. 2023) and monitoring programs are expected to be part of the environmental impact assessment in order to control that the mitigation measures are efficient enough to ensure the "no net loss" of biodiversity (European Parliament 2014). Such a framework paves the way for the integration of biodiversity-oriented monitoring systems into the TI and its vicinity in line with proposals done for hydraulic management (Wang et al. 2022) or user safety (Proto et al. 2010).

If using existing cameras around the TI or embedding ones dedicated to biodiversity monitoring may contribute to map the AVC risk, their deployment must be optimised to ensure the system cost efficiency as well as its sustainability (Hautière et al. 2012, 2023). In this respect, literature issuing from sensor-based biodiversity monitoring systems provides recommendations (e.g. distance between devices, recording frequencies, etc) (Evans et al. 2019; Kays et al. 2020; Nawaz et al. 2021). Unfortunately, these recommendations are often hardly applicable to the survey of linear structures such as roads, railways or channels. However, based on the three first steps of the proposed framework, scenarios of sensors network deployment can be tested and ultimately optimised by automatically removing or adding devices in the sensor network.

Developing performant artificial intelligence to recognise species involved in AVCs

The recognition algorithm fine-tuned in this work is not general enough to properly perform in operative conditions. The moderate performances of the model are due to multiple factors such as the number of annotated data used to train the model and particularly the lack of pictures taken in operative-like conditions. To improve these performances, we successfully used DeepFaune which was trained on larger data set to recognise our focal species among other French common ones (Rigoudy et al. 2022). Albeit the marginal performance improvement on the data from the showcase, its use in other places of the general monitoring program shows very poor performances, for instance when cameras are elevated and animals for which only the back can be seen. To address these current limitations, further recognition algorithms developed to ultimately map AVC should focus on a limited number of relevant species and on the deployment conditions (e.g. sensor orientation, image quality, etc.). In addition, the use of deep learning to recognise species leads to changes in the form of the abundance model inputs (false positives, uncertainty in the recognition, etc.). Further research in the domain of statistical analysis of ecological data may adapt to this new form of input data (Chambert et al. 2018; Tabak et al. 2020) and may help in overcoming the current limited performance of recognition algorithms to ultimately produce an AVC risk map.

From a static map of avc risk to real time driver information

As sensors collect data continuously, our framework could possibly be improved by using abundance or occupancy models in continuous-time (Guillera-Arroita et al. 2012, 2012). Continuous-time data discretised do not respect the mathematical hypothesis of classical discrete-time models, as sampling occasions are not temporally independent (Barbour et al. 2013). A continuous-time model would make our framework more objective and reproducible, as the discretisation period is chosen arbitrarily (Rovero and Zimmermann 2016; Schofield et al. 2017; Rushing 2023), as well as the time interval in which images are removed because they are likely to be the same individual, and would avoid losing information (Kellner et al. 2022). Continuous-time models have recently been developed for unmarked populations (for example Guillera-Arroita et al. (2011) for occupancy, Guillera-Arroita et al. (2012) for abundance, and even Kellner et al. (2022) for co-occurrence), which could be useful for collisions-involved species whose distribution is strongly linked to other species (Hebblewhite 2007; Rioux et al. 2022). This framework would also improve with the development of incremental learning (Zhu et al. 2022), to produce dynamic adaptive maps that could be ultimately sent to connected vehicles.

Mainstreaming biodiversity in the digital twins of transport infrastructure

Digital twins are developing regardless of the TI type (e.g. road, railway, airport, etc) and the framework we proposed can be applied to any type of TI with, for instance, some adaptation for bird detection in a 3D explicit digital environment to manage collisions with planes (Dziak et al. 2022). Similar approaches are also being designed for the development of smart cities and territories (Catalano et al. 2021). Generalising biodiversity monitoring integration in the interconnected digital twins of the built environment offers a great opportunity to contribute to the survey of biodiversity global trends as a co-benefit of the ongoing digitalisation of landscape management (ANZLIC 2019; Singh et al. 2021; Moulherat et al. 2022).

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

SM and LP mainly wrote the manuscript, SM performed the simulations, LP, GD, GT, JPT trained the deep learning algorithms, SM, LP, MPE and OG performed the analysis, LG digitised the land cover, SM, NH, JPT and OG conceptualised the paper, SM, JPT and OG obtained the grant. All the authors contributed to the paper edition and approved the final version.

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Data availability

All of the data that support the findings of this study are available in the main text or Supplementary Information.

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Supplementary material 1

Complementary details of the methods used in the paper as well as additional results

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Data type: docx

Explanation note: All the source data and code that can be shared and are available on an online repository at https://oikolab.terroiko.fr/publications/monitoring-and-animal-vehicle-collisions-nat-conserv-2024 . For data with restricted access, the supplementary material explains how to access them on demand.

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Research Article

The potential of electrified barriers to keep black bears out of fenced road corridors at low volume access roads

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Abstract

Fences can reduce wildlife-vehicle collisions, but it is not always possible to fence over long distances, especially not in multi-functional landscapes. Side roads, driveways, and the need for access to agricultural fields all result in gaps in the fence. In some cases, wildlife guards or gates are installed at access points. However, gates usually require people to get in and out of their vehicle and they are often left open. Wildlife guards are typically only suited for low traffic speed, and while they can be a substantial barrier to ungulates, they are readily crossed by species with paws, including bears. Electrified barriers embedded in travel lanes can be a substantial barrier to both ungulates and bear species and while they can be suitable for higher traffic volume and speed, the costs are typically higher than for low volume and low speed roads. We explored the potential of low-cost electrified barriers to keep bears from accessing fenced road corridors at low traffic volume and low speed vehicle access points. As a first step, we conducted the study on private land at a melon patch that was a known attractant for black bears. We investigated the effectiveness of an electric fence and 5 different types of electrified barriers designed to keep black bears out of the melon patch. The electrified barriers included a swing gate, a standard bump-gate, a modified bump-gate with conductive netting, drive-over wires a few inches above the ground, and a drive-over mat. Trail cameras were installed at each access point to document approaching black bears and potential crossings into the melon patch. The swing gate, modified bump-gate, drive-over wires, and drive-over mat were an absolute (100%) or near absolute barrier (94.3%) for black bears while the standard bump-gate was a poor barrier (48.4%). Through a step-by-step process, the weak points of the electrified barriers at the vehicle access points and the electric fence around the melon patch were addressed. After addressing a weak point at a vehicle access point, the bears increasingly dug under the fence to enter the melon patch. However, eventually the melon patch became almost inaccessible to black bears. The number of black bears trying and succeeding to enter the melon patch at a particular location depended on how difficult it was to enter at other locations. This illustrates that fences and vehicle access points should be designed, operated, maintained, and monitored as a system rather than as individual features, regardless of whether the goal is to protect crops or to keep animals out of a fenced road corridor. The total number of black bear observations at the locations monitored with a trail camera, regardless of which side of the fence or electrified barriers the bears were on, was 95% lower in 2021 than in 2020. Combined with having no indication of a substantial drop in black bear population size from 2020 to 2021, this suggests that after the black bears were no longer able to enter the melon patch, they drastically reduced their presence in the immediate surroundings and reduced their effort to try and access the crop; the attraction of the melon patch and the habit of eating its melons was broken.

Key words: Access, coexistence, collision, conflict, crop, drive, fence, gate, highway, human, interaction, mitigation, roadkill, vehicle, wildlife

Introduction

Most wildlife mitigation measures along highways are aimed at improving human safety, reducing direct wildlife mortality, and providing safe crossing opportunities for wildlife (e.g. Ford et al. 2009; van der Grift et al. 2017). Fences in combination with wildlife crossing structures are probably the most effective combination of mitigation measures to achieve these objectives (Clevenger and Waltho 2000; Rytwinski et al. 2016; Huijser et al. 2021). For fences to be reliably reducing collisions with large wild mammals by 80% or more, at least 5 kilometers of road length needs to be fenced, including a buffer zone that extends well beyond the known hotspots for wildlife-vehicle collisions (Huijser et al. 2015; Huijser et al. 2016a). Collisions that still occur within or adjacent to the fenced road sections tend to be concentrated near the fence-ends (Huijser et al. 2016b; Plante et al. 2019; Huijser and Begley 2022). In addition, gaps in fences at access roads can result in concentrations of collisions inside fenced road sections (Sawyer et al. 2012; Cserkész et al. 2013; Yamashita et al. 2021).

Embedding barriers (e.g. wildlife guards or electrified barriers) in the travel lanes at fence-ends or at access roads can reduce intrusions into the fenced road corridor (Peterson et al. 2003; Gagnon et al. 2019; Honda et al. 2020). Gates are commonly used at gaps in the fence at low traffic volume access roads, but they are often left open, allowing wildlife to access the road corridor (VerCauteren et al. 2009; Sawyer et al. 2012). While single wide cattle guards or wildlife guards (2.1–3.0 m) can be effective for some ungulate species (VerCauteren et al. 2009; Huijser et al. 2015; Honda et al. 2020), double wide cattle or wildlife guards (4.6–6.6 m) consisting of round bars or bridge grate material and situated above a pit, are generally recommended for ungulates (Belant et al. 1998; Peterson et al. 2003; Allen et al. 2013; Cramer and Flower 2017; Gagnon et al. 2020; Kintsch et al. 2021). However, some designs, including guards that consist of flat bars, are less effective for ungulates (Reed et al. 1974; Kintsch et al. 2021), and single or double wide guards are not a substantial barrier for species with paws, including many mid-sized and large carnivore species (Allen et al. 2013; Clevenger and Barrueto 2014; Huijser et al. 2015, 2016b; Honda et al. 2020). Electrified mats or electrified guards can be a barrier for both ungulates and species with paws, but to prevent animals from jumping across the mat, they may need to be 4.6–6.6 m wide (Seamans and Helon 2008; Cramer and Flower 2017). Combinations of electrified barriers and non-electrified guards are also possible (Gagnon et al. 2020).

We explored the potential of low-cost electrified barriers to keep bears from accessing fenced road corridors at low traffic volume and low speed vehicle access points. As a first step, we conducted the study on private land at a melon patch that was a known attractant for black bears (*Ursus americanus*). We investigated the barrier effect of an electric fence and different types of electrified barriers at vehicle access points in keeping the bears out of the melon patch. In the past, the farmer has seen up to 7 individual black bears eating melons in the patch at the same time (personal communication Cassie Silvernale). In 2019, before the electrified fence and barriers were put in place, the economic losses because of black bears were estimated at 5% of the crop or about 5,000 melons (Andrews 2020). Assuming a sale price of about US $5 per melon, this amounts to about US $25,000 in lost revenue per year. Eating melons with high sugar and carbon content can also be detrimental to the health of the black bears, e.g. through tooth decay. Furthermore, reducing conflict between farmers and bears in general can help build a willingness to co-exist in the same landscape (e.g. Wilson et al. 2017). Depending on the location, this may benefit both black bears and grizzly bears (*Ursus arctos*), as well as other species that are present in the area and that may eat commercial crops.

Methods

Study area

The main study area was a melon patch (about 8 ha) located immediately south of the Bison Range, about 3.5 km west of Ravalli, Flathead Indian Reservation, Montana, USA (Fig. 1). The melon patch was just north of MT Hwy 200, and just south of the Jocko River and the associated trees and shrubs in the riparian area. While the patch was dominated by different varieties of melons, there was also some corn planted along one edge of the field. An additional study site with a drive-over mat was located along US Hwy 93, just south of Ravalli (Fig. 1).

The electric fence and electrified barriers at the vehicle access points

A Non-Governmental Organization, People and Carnivores, built an electric fence around the melon patch in the summer of 2020 (Fig. 2). The fence was constructed before the harvest of the melons in 2020 with the intent of keeping black bears, and potentially also grizzly bears, out of the melon patch. In addition, electrified barriers were installed at 4 vehicle access points to the melon

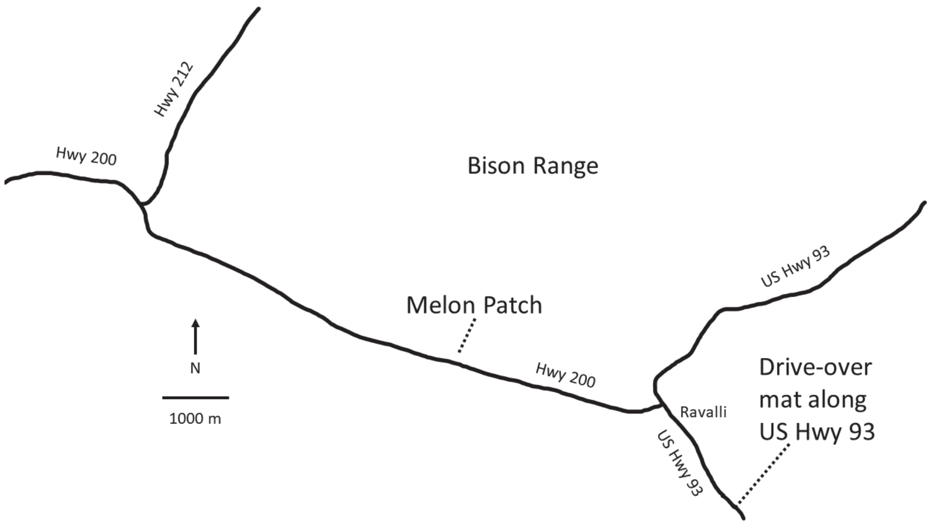

Figure 1. The location of the melon patch, and the additional drive-over mat, Flathead Indian Reservation, Montana, USA.

patch (Fig. 2). The electric fence consisted of 4 wires attached to composite fence posts made from polypropylene and wood (PasturePro®) (Fig. 3). This fence design is similar to those used by others to keep different bear species from accessing crops or other attractants (Huygens and Hayashi 1999; Otto and Roloff 2015; Khorozyan and Waltert 2020). Corner posts and braces, including at vehicle access points, were treated wood posts. The height of the fence was about 91 cm (Fig. 3). While the fence was designed to keep both black bears and grizzly bears out of the melon patch, it was not designed to keep other species from accessing the melon patch. Ungulates (e.g. white-tailed deer (*Odocoileus virginianus*), mule deer (*O. hemionus*), and elk (*Cervus canadensis*)) can easily jump this fence. The farmer sprayed herbicides along

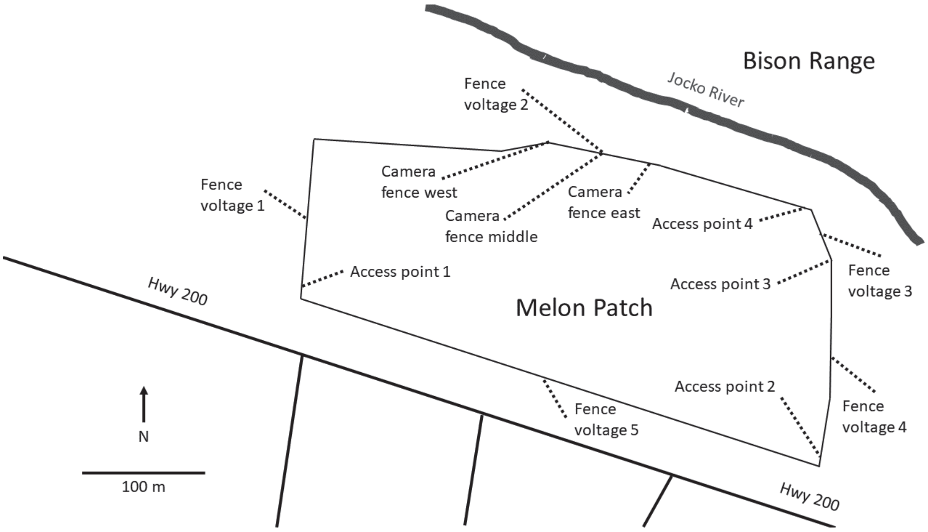

Figure 2. The melon patch (roughly 450 m east-west and 180 m north-south). The electric fence (solid black line around the melon patch), the 4 vehicle access points, the 5 locations along the fence where fence peak DC voltage was measured, and the location of 3 trail cameras where we suspected black bears were digging under the fence.

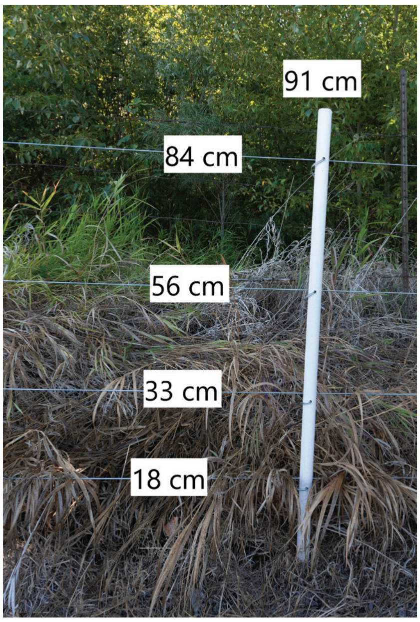

Figure 3. The approximate height of the non-modified electric fence and its 4 wires.

the fence to reduce voltage drop. At first, the 2nd wire from the bottom was a designated ground wire, with the other 3 wires being hot (i.e., carrying current). After 27 August 2020, the 2nd wire from the ground was made into a hot wire also, which meant that from then on, the current received by an animal depended on the contact points of the animal with the ground which varies with the conductivity of the animal itself, and the conductivity of the ground or vegetation. A solar panel, and associated battery and energizer powered the fence and all 4 access points. However, from 6 August 2021 onwards, access point 3 had the drive-over mat installed which was powered by its own solar panel and associated battery and energizer. The electric fence energizers are listed by the internationally recognized safety standards agencies (UL, CSA or IEC).

In 2020 and 2021, we evaluated 5 different electrified barrier designs at the 4 access points to the melon patch (Table 1, Fig. 4). The drive-through bump-gates were originally installed at 2 access points, but 1 of them was modified in 2020, and the drive-over wires barrier was replaced by a drive-over mat in 2021 (for exact dates see Table 1). In 2022, we continued to monitor only the swing gate at access point 2 and the drive-over mat at access point 3 (Table 1). We continued monitoring in 2022 to increase the sample size for the swing gate (from 9 in 2020-2021 to 23 in 2020-2022) and the drive-over mat (from 2 in 2021 to 3 in 2021-2022). Since the drive-over mat still had a very low sample size, data from a similar barrier design from the same manufacturer at a nearby site (US Hwy 93, just south of Ravalli) was added (Fig. 1). The electrified barrier along US Hwy 93 was monitored in 2022 and 2023 (Table 1). There were 2 black bears and 1 grizzly bear that approached this drive-over mat, bringing the total sample size for the drive-over mat design to 5 black bears and 1 grizzly bear (6 "bears" in total).

The fence and the 5 electrified barriers at the 4 access points to the melon patch were modified during the study. The most important modifications to the electrified barriers, start and end dates of the melon picking seasons in 2020 and 2021, and the associated evaluation periods, are summarized in Table 2. Note that melons were also a potential attraction to animals before the first pick of a season and after the last pick of a season.

Table 1. The electrified barriers and the periods during which they were evaluated.

Electrified barrier type	Brand, approximate costs (US$)	Location	Evaluation start-end 2020	Evaluation start-end 2021	Evaluation start-end 2022	Evaluation start-end 2023
Swing gate (modified with 4 hot wires)	Hutchison, $290 for gate only (excl. installation)	Access point 2	10 Jul – 12 Dec	28 Apr – 19 Nov	23 May – 15 Dec	None
Bump-gate (not modified)	Koehn, $180, (excl. installation)	Access point 1	10 Jul – 12 Dec	28 Apr – 19 Nov	None	None
Bump-gate (not modified)	Koehn, $180 (excl. installation)	Access point 4	10 Jul – 27 Aug	None	None	
Bump-gate (modified with netting)	Koehn, $180 (excl. installation, excl. netting)	Access point 4	27 Aug – 12 Dec	28 Apr – 19 Nov	None	None
Drive-over wires	Fully custom (Bryce Andrews, People and Carnivores), cost under $500 (excl. installation)	Access point 3	10 Jul – 12 Dec	28 Apr – 4 Aug	None	None
Drive-over mat	Crosstek™, $11,250 (incl. installation)	Access point 3	None	6 Aug – 19 Nov	23 May – 15 Dec	None
		US Hwy 93	None	None	23 May – 30 Nov	14 Apr – 12 Jul

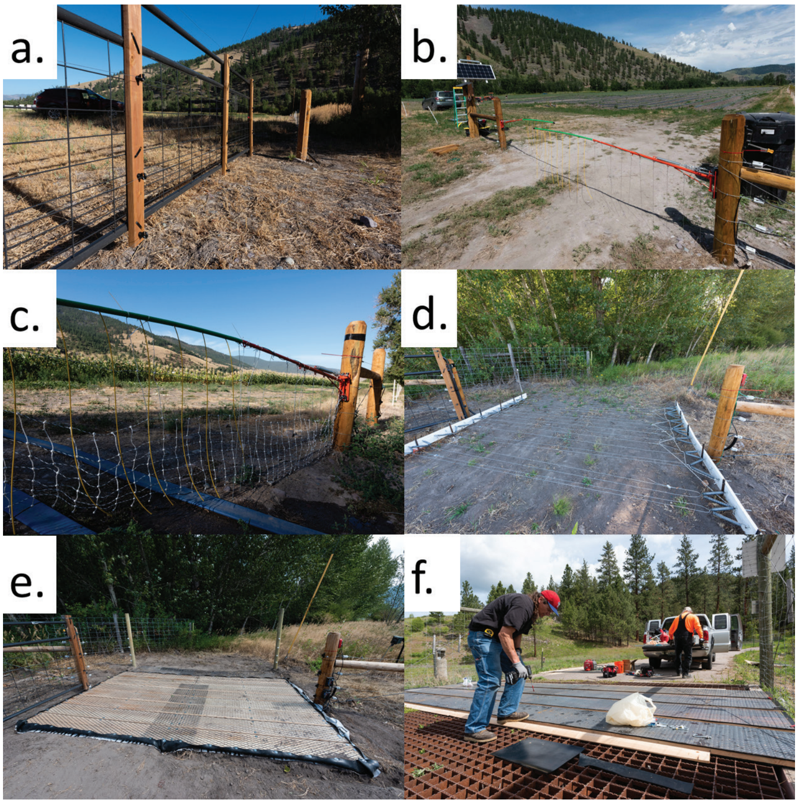

Figure 4. The 5 electrified barriers at the 4 vehicle access points (**a–e**) and along US Hwy 93 (**f**) **a** The electrified swing gate (4.88 m wide, 1.37 m tall, about 18 cm gap between ground and bottom of gate). The wires are mounted at 15, 48, 97, and 140 cm above the ground. Installed at access point 2 **b** A drive-through bump-gate (not modified) (about 4.88 m wide, about 91 cm tall), with vertical electrified wires. The orange horizontal pole is metal and carries current. The green horizontal part is fiberglass and does not carry current. Installed at access points 1 and 4 **c** A modified drive-through bump-gate (with conductive netting) (about 4.88 m wide, about 91 cm tall), with vertical electrified wires and custom conductive netting (about 61 cm high) attached. Installed at access point 4 **d** The drive-over wires, about 4.34 m wide (post-post) and 3.10 m long. The 18 drive-over wires are about 18 cm above the ground and the gaps between the wires vary between 13-30 cm. There are "side-board" wires that angle toward the ground from the post along the sides of the barrier to reduce the likelihood of an animal bypassing the drive-over wires. However, these "side-boards" do not cover the full length of the barrier. Installed at access point 3 **e** The drive-over mat, about 4.34 m wide (post-post) and about 3.05 m long. On the far side, the habitat side, there is metal mesh on the ground, connected to a grounding rod (about 61 cm wide). This is followed by 8 sections of 25-28 cm wide expanded metal sheeting (alternating positive and negative (ground)) mounted on wooden planks. This drive-over mat is powered by its own solar panel, battery and energizer. Installed at access point 3 **f** The drive-over mat, about 7.36 m wide (post-post) and about 2.44 m long, installed at a gap in a wildlife fence for a low volume access road along US Hwy 93. The mat is positioned on a wildlife guard (bridge grate material). On the far side, the habitat side, the animals first encounter bridge grate material (about 66 cm wide) that is connected to a grounding rod. This is followed by a wooden ramp (about 20 cm wide) and 4 metal plates (each about out 51 cm wide, alternating positive and negative (ground)) mounted on rubber and wooden planks with about 6 cm gaps in between the metal plates, and another wooden ramp on the far side (about 20 cm wide). This drive-over mat is powered by its own solar panel, battery and energizer. Installed along US Hwy 93, just south of Ravalli (see Fig. 1).

Data collection

Each access point had a trail camera installed (Reconyx™ PC 900). The trail cameras fully covered the area up to 2 m in front of each access point. The 2 m distance from the access point was visible on each image based on the line between the trail camera's viewpoint and a stick with reflective tape on the other end. This allowed us to consistently evaluate the behavior of large mammals that approached each access point within 2 m. We evaluated whether the animals succeeded in accessing the melon patch by crossing the electrified barriers. Some large mammals were also detected further away from the access point, but those animals were not included in the evaluation of the effectiveness of the electrified barriers at the 4 access points. We also detected or suspected that bears were digging under the electrified fence at 3 locations (Table 3, Fig. 2), and we monitored these locations with trail cameras. Interestingly, these locations were all adjacent to the riparian habitat along the Jocko River and not along the roadside of MT Hwy 200 or adjacent agricultural fields. We replaced the memory cards in all trail cameras about once a month. We replaced the batteries (Energizer® Ultimate Lithium™) about every 3 months.

We measured the peak DC voltage (Stafix digital volt meter) of the electrified fence on each of the 4 wires at 5 locations for most of the trail camera checks before, during, and after the melon harvest seasons in 2020 and 2021 (Figs 2, 3). We also measured the peak DC voltage of the electrified barriers at the 4 access points during most trail camera checks before, during, and after the harvest seasons in 2020 and 2021 (Figs 2, 4). For the bump-gate designs at

Table 2. Major modifications to the electrified fence and vehicle access points and the start and end dates of the melon picking seasons in 2020 and 2021. The number of days relates to the length of each period with a particular set of conditions.

From	Until	Days (N)	Description of changes that applied to the period
10-Jul-20	7-Aug-20	28	Electricity turned "on" 10 Jul 2020, turned "off" 12 Dec 2020
7-Aug-20	21-Aug-20	14	Start melon picking season 7 Aug 2020, end 2 Oct 2020
21-Aug-20	27-Aug-20	6	Wires lowered a select locations, access points permanently "on"
27-Aug-20	9-Sep-20	13	Mesh added at access point 4, motion light fence west, 2nd wire from bottom hot
9-Sep-20	2-Oct-20	71	Additional post and a 5th wire at fence west and fence middle
2-Oct-20	12-Dec-20	23	No more melon picking for 2020 season
28-Apr-21	30-Jul-21	93	Electricity turned "on" 28 Apr 2021, turned "off" 19 Nov 2021
30-Jul-21	6-Aug-21	7	Start melon picking season 30 Jul 2021, end 14 Sep 2021
6-Aug-21	14-Sep-21	39	New drive-over barrier installed at access point 3 on 6 Aug 2021
14-Sep-21	19-Nov-21	66	No more melon picking for 2021 season

Table 3. The 3 fence locations (see Fig. 2) and the periods and associated number of days they were monitored with a trail camera in 2020 and 2021.

Fence location	2020		2021	
	Start and end date evaluation	Days (N)	Start and end date evaluation	Days (N)
West	6 Jul 2020 - 12 Dec 2020	159	28 Apr 2021 - 19 Nov 2021	205
Middle	27 Aug 2020 - 12 Dec 2020	107	28 Apr 2021 - 19 Nov 2021	205
East	6 Jul 2020 - 12 Dec 2020	159	28 Apr 2021 - 19 Nov 2021	205

access points 1 and 4, the peak DC voltage was measured for both the right and the left part of the gate. For the other barriers there was just 1 measurement. Though the peak DC voltage measurements only occurred during the trail camera checks - the voltage measurements were not continuous - the peak DC voltage readings showed whether the electric fence and electrified barriers were electrified at that moment. This indicated whether we could have expected the electric fence and electrified barriers at the access points to have discouraged bears from entering the melon patch since the last voltage readings.

Data analyses

Voltage

For most of the trail camera checks during the melon harvest seasons in 2020 and 2021, we calculated the average peak DC voltage for each of the 4 wires of the fence based on the 5 measurement locations. In addition, we calculated the average peak DC voltage for each of the 2 sides of the 2 bump-gates at access points 1 and 4. The peak DC voltage of the barriers at access points 2 and 3 was always based on a single measurement.

Barrier effect

Each barrier design at a vehicle access point was evaluated for its barrier effect on black bears through counting the number of black bears that were recorded in the area up to 2 m immediately in front of each access point and calculating the percentage of bears that were deterred. If a black bear was recorded within 5 minutes of the previous event involving a black bear, it was considered the same bear and it was counted and evaluated as 1 event. However, if there was evidence (e.g. based on body size, hair color) that these were different individuals, then it resulted in 2 events. If more than 5 minutes passed between consecutive black bear observations, then these were considered different events, regardless of whether it involved the same bear. We reviewed the images and calculated the percentage of black bears that successfully accessed the melon patch (undesired result) vs. the percentage of black bears that were deterred (desired result, equivalent to the barrier percentage).

Breaking the addiction

Modifications to the barriers at the vehicle access points and the fence were recorded and grouped into different periods (Table 2). The absolute number of black bears accessing the melon patch in each period was calculated for each access point and for each of the 3 fence locations that were monitored with a trail camera. Since the periods varied greatly in length, the black bear counts were standardized for the number of days in each period. This analysis shows potential increase or decrease in black bears accessing the melon patch, and the locations of the intrusions in association with the modifications to the barriers at the vehicle access points and the fence. In addition, we counted the total number of black bear events inside and outside the fenced melon patch recorded by the 7 trail cameras, regardless of the distance to the fence, vehicle access point or trail camera, per month for both 2020 and 2021. This analysis showed potential differences in the attraction of the melon patch and whether the black bears' habit of trying to access the melons was broken.

Results

Voltage

When measured, the peak DC voltage on the fence was almost always 7–9 kV and very similar for the 4 fence wires (Fig. 5). The 2nd wire from the bottom was a ground wire until 27 August 2020, hence the lack of voltage measurements before that date. In general, the 2 bump-gates (access points 1 and 4) had lower peak DC voltage (usually between 4–6 kV) than the barriers at the other 2 locations (usually between 7–10 kV) (Fig. 6). Note that the measurements in 2021 at access point 3 related to a drive-over mat with its own power source that was independent from that of the fence and the 3 other access points. Also note that the peak DC voltage at the end of November in 2021 was lower everywhere.

Barrier effect access points

All the bears that were recorded at the melon patch were black bears; there were no observations of grizzly bears at this site. All the black bear events related to single animals; there were no events involving multiple black bears (e.g. a sow and cubs). Four out of the 5 electrified barrier designs for access points were an absolute (100%) or near absolute barrier (94.3%) for black bears (Fig. 7). However, the bump-gates that were originally designed for cattle were a poor barrier for black bears (48.4%). Based on the images from the trail cameras, the bears usually passed in between the vertical electrified strands, and thus minimized contact with the wires. After conductive netting was attached to the bump-gate at access point 4, the bears no longer passed through that bump-gate (100% barrier) and

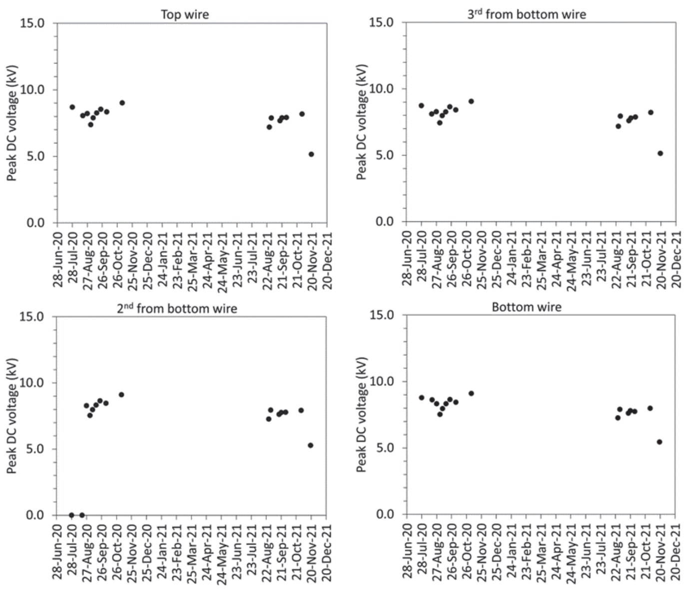

Figure 5. The peak DC voltage on the 4 fence wires before, during, and after the harvest melon seasons in 2020 (point groups on the left) and 2021 (point groups on the right).

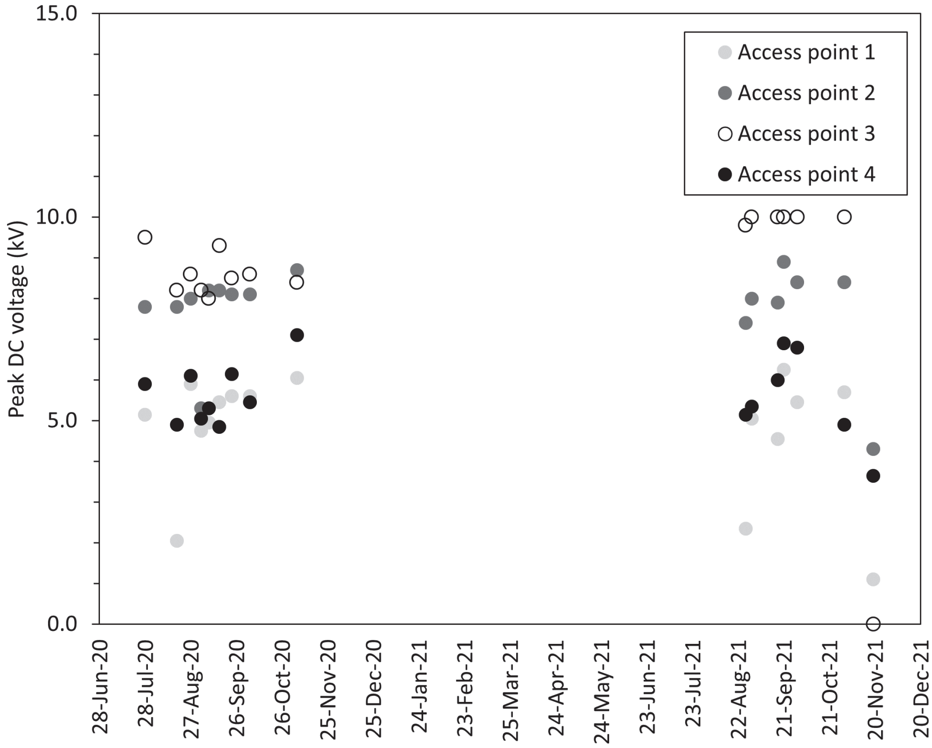

Figure 6. The peak DC voltage at the 4 access points before, during, and after the melon harvest seasons in 2020 (point groups on the left) and 2021 (point groups on the right). The measurements in 2021 at access point 3 related to a drive-over mat with its own power source that was independent from that of the fence and the 3 other access points.

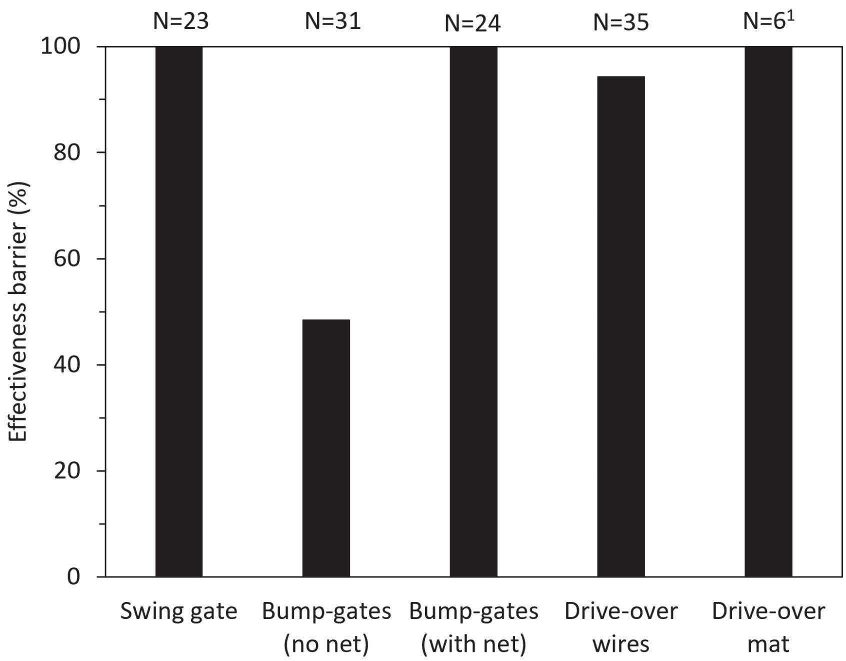

Figure 7. The effectiveness of the different barriers in keeping black bears out of the melon patch. N = number of bears (sample size) approaching the barriers. [1]Sample size at melon patch was 3 black bears, but the data were supplemented by additional observations (2 black bears and 1 grizzly bear, all single animals) at a similar mat nearby (US Hwy 93, south of Ravalli, see Fig. 1).

moreover, they did not even touch the electrified barrier. The same was true for the swing gate, drive-over wires, and drive-over mat; most bears that approached the barriers did not touch the electrified components at all. Two out of 35 black bears passed the drive-over wires above the ground (Fig. 7). Both intrusions involved a bear bypassing most of the wires by coming in or leaving from the side.

Breaking the addiction

At the 7 locations that were monitored with a trail camera, the number of black bear intrusions into the melon patch varied between 2.07 and 4.38 per day during the 2020 melon picking season (103 intrusions between 7 August 2020 and 2 October 2020) and was 0 per day during the 2021 melon picking season (0 intrusions between 30 July 2021 and 14 September 2021) (Fig. 8). Access point 4, the non-modified bump-gate adjacent to the Jocko River, and "fence west" accounted for most of the intrusions. After conductive netting was attached to the bump-gate at access point 4, black bear entries increased at "fence west" and at "fence middle" (Fig. 8). After fence modifications on 9 September 2020, almost no bears were able to enter the melon patch. There were only 2 black bears that entered after that date, and both did so through partially bypassing the drive-over wires at access point 3. We did not have any evidence that black bears entered the melon patch at any other location that was not monitored with a trail camera. The total number of black bears was highest in August through October (Fig. 9). The total number of black bear observations at any of the 7 locations monitored with a trail camera, regardless of the distance to the fence, vehicle access point or trail camera, and regardless of which side of the fence or electrified barriers the bears were on, was 95% lower in 2021 (N=24) than in 2020 (N=527).

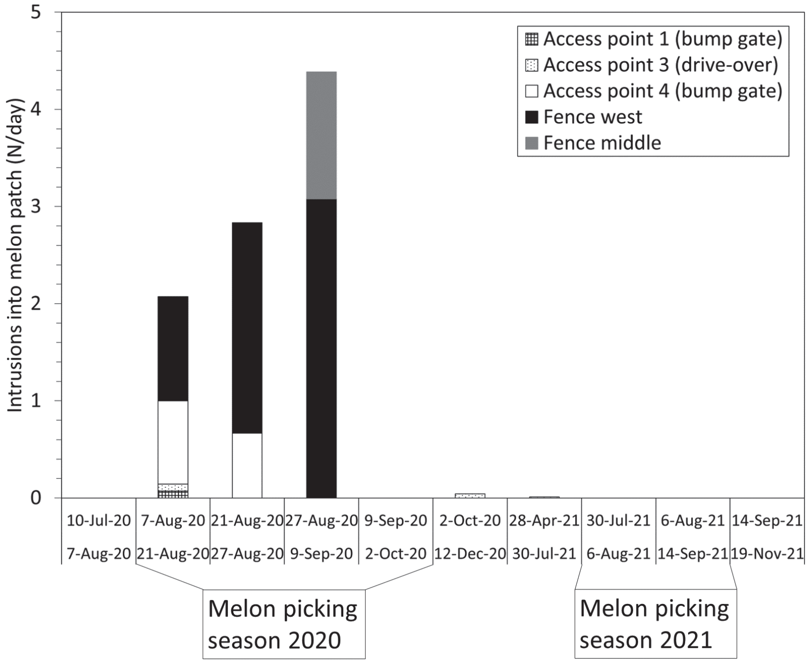

Figure 8. Black bear intrusions into the melon patch during different periods in 2020 and 2021. Note: No intrusions were reported at access point 2 (swing gate) and "fence east". Note: Access point 3 had drive-over wires 10 July 2020 until 6 August 2021 and a drive-over mat after 6 August 2021.

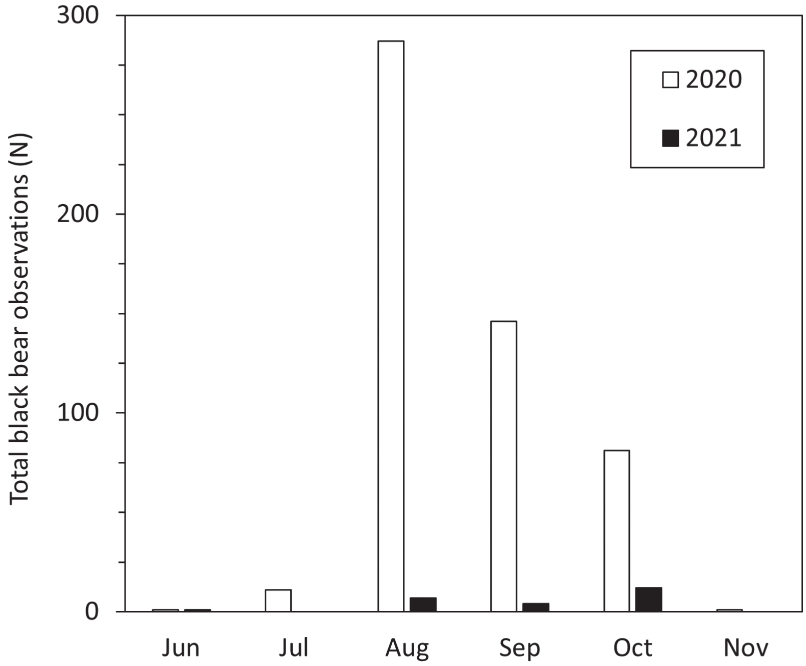

Figure 9. The total number of black bear observations, regardless of the distance to the fence, vehicle access point or trail camera, and regardless of which side of the fence or electrified barriers the bears were on, per month in 2020 and 2021.

Discussion

Voltage

In general, the peak DC voltage on the fence and at access points 2 and 3 was almost always 7–10 kV. However, the 2 bump-gates (access points 1 and 4) usually had lower peak DC voltage (usually between 4–6 kV), suggesting higher resistance of the materials or a short or voltage leak. The overall drop in peak DC voltage at the end of November in 2021 was most likely the result of moisture causing a voltage leak, or shorter days (not enough daylight to recharge the batteries) and lower temperatures (reduced capacity of the batteries). It is possible that there were additional voltage drops in between voltage readings, but if they were present at all, the peak DC voltage had recovered by the next voltage measurement.

Barrier effect access points

Four out of the 5 electrified barrier designs for access points were an absolute or near absolute barrier for black bears. However, non-modified bump-gates that were originally designed for cattle were a poor barrier for black bears as they deterred only about half of the animals. Adding conductive netting to one of the bump gates made it into an absolute barrier, however. This is likely because the netting results in more contact points with an animal for a longer time when an animal tries to lift the fence material and pass under. Changing electrified wires to electrified netting also made a fence a much greater barrier to European wild rabbits (*Oryctolagus cuniculus*) (McKillop et al. 1992). The drive-over wires were partially bypassed by 2 bears, emphasizing the need for tight fences along the full length of the sides of the barrier. Interestingly, most bears that approached an absolute or near absolute barrier did not even touch the barrier. This is consistent with other studies that reported that black bears and other large mammal species tend to stay away from electrified barriers, apparently because they know about its potential impact (Huygens and Hayashi 1999; Fischer et al. 2011; Otto and Roloff 2015; Teixeira et al. 2017).

Breaking the addiction

Through a step-by-step process, the weak points of the electrified barriers at the vehicle access points and the electric fence around the melon patch were addressed. The standard bump-gate at access point 4, and 2 fence locations (west and middle) were of particular concern. Interestingly, once the conductive netting was attached to the bump-gate at vehicle access point 4, the bears increasingly dug under the fence to enter the melon patch. However, from 9 September 2020 onwards, including during the melon picking season in 2021, the melon patch became almost inaccessible to black bears; the number of intrusions by black bear was reduced from 2.07–4.5 per day (2020 melon picking season) to 0 per day (2021 melon picking season). We found that black bears trying and succeeding to enter the melon patch at a particular location depended on how difficult it was to enter at other locations. This illustrates that fences and vehicle access points should be designed, operated, maintained, and monitored as a system rather than as individual features, regardless of whether the goal is to protect crops or to keep animals out of a fenced road corridor.

The total number of black bear observations at any of the 7 locations monitored with a trail camera, regardless of which side of the fence or electrified barriers the bears were on, was 95% lower in 2021 than in 2020. The farmer reported substantial reductions in melons lost to bears in 2021 compared to 2019 before the fence and electrified barriers were installed (100% reduction in losses; personal communication Cassie Silvernale). Although black bear activity in and adjacent to the melon patch was still relatively high in 2020, not nearly as much melon loss occurred because of bears in 2020 compared to 2019 (80% reduction in losses; personal communication Cassie Silvernale). The very substantial reductions in both black bear presence and melon losses after the installation of the fence and electrified barriers suggests that after the black bears were no longer able to enter the melon patch, they drastically reduced their presence and effort to try to access the melon patch. Apparently, the attraction of the melon patch and the associated habit of eating melons was broken. Note that based on direct observations of black bears by the authors, there was no indication of a population crash of black bears in 2021 in the immediate vicinity, or in the wider region (Montana Fish Wildlife & Parks 2024).

Experiences with operation and maintenance

All 4 vehicle access points had manual switches allowing farm personnel to walk through the electrified barriers without shocking themselves. However, shortly after the electric barriers were activated, the farmer realized that the switches were sometimes accidentally left in the "off" position. From 29 August 2020 onwards, all switches were permanently taped in the "on" position. While relatively inexpensive, the bump-gates required custom conductive netting to become a substantial barrier to black bears. The netting is subject to tearing and needs to be adjusted and reattached regularly (e.g. with zip-ties). In addition, there are tensioners for the 2 horizontal poles of the bump-gates. These also need to be adjusted on a regular basis to ensure that the 2 horizontal poles align and do not leave a gap in the middle. The horizontal poles are also subject to breaking; one of the poles broke after it got stuck in a bumper or wheel well of a pickup truck.

There were no operation or maintenance issues with the electrified swing gate. A design problem of the drive-over wires barrier was that the side fences were too short. Both intrusions by black bears involved the animals bypassing most of the wires above the ground by accessing or leaving the barrier from one of the sides. Barriers or "side-fences" that run tight along the full length of the barrier would likely address this issue and force bears, if they try to access the crop, to walk on top of, or in between, the wires above the ground for the full length of the barrier. In contrast, the drive-over mat has full side barriers that have a tight connection to the mat, and they do run the full length of the mat. Here, no large mammals were able to bypass the mat by coming in from or leaving at one of the sides. We did observe that the drive-over mat can kill amphibians and small mammals. Between 6 August 2021 and 21 November 2021, we found 1 dead western toad (*Anaxyrus boreas*) and 1 dead deer mouse (*Peromyscus maniculatus*) on the mat. Such unintended side effects may be reduced through making the habitat immediately adjacent to the barrier less attractive or inaccessible to small species, e.g., through ABS screens attached to the side fences of the barriers. There could also be a sensor installed that would only turn the electricity on after a large animal has been detected that is approaching the barrier from the habitat side.

For locations that are accessible to the public, such as along highways, warning signs and "turn electricity off" buttons may be required. These buttons should be associated with a timer and an indicator light so that the electricity will automatically turn on again, e.g. after a minute or so, and that confirmation is visible to the public. Note that some of the tested electrified barrier designs are not a suitable barrier to ungulates (e.g. swing gate, bump-gates). Wider barriers such as the drive-over mat or, though to a lesser extent, drive-over wires, are or can be, at least in theory, substantial barriers to not only species with paws but also ungulates (Fischer et al. 2011; Gagnon et al. 2019).

Conclusion

After modifications, a combination of an electric fence and electrified barriers at vehicle access points was able to keep almost all black bears out of a melon patch and break their habit of eating melons in the melon patch. However, bump-gates required custom conductive netting and frequent adjustments and repairs. The electrified swing gate was an absolute barrier to black bears and had no maintenance issues. However, this design still requires people to get in and out of their vehicle when opening and closing the gate, and therefore the gate may be left open. The drive-over wires barrier was a near absolute barrier for black bears. Nonetheless, its effectiveness can likely be improved if the side barriers run tight along the full length of the barrier. The drive-over mat performed well but has only a small sample size. The downside of the drive-over mat, and possibly also of the drive-over wires, is that these types of electrified barriers may occasionally kill small animal species (e.g. amphibians, reptiles, small mammals). Although the effectiveness of these barriers was investigated at a melon patch on private land, the results are applicable to low traffic volume and low traffic speed access points along fenced public highways. These electrified barriers are especially important along road sections where the purpose of wildlife fences is to also keep species with paws out of the fenced road corridor.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

Conceptualization: MPH. Data curation: SCG, MPH. Formal analysis: MPH. Funding acquisition: MPH. Investigation: MPH, SCG. Methodology: SCG, MPH. Project administration: MPH. Supervision: MPH. Validation: MPH, SCG. Visualization: MPH. Writing - original draft: MPH. Writing - review and editing: SCG, MPH.

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Data availability

All of the data that support the findings of this study are available in the main text or Supplementary Information.

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Supplementary material 1

Barrier effect of electrified barriers to bears at the melon patch and the additional site with drive over mat along US Hwy 93, Montana, USA

Authors: Marcel P. Huijser, Samantha C. Getty

Data type: xlsx

Explanation note: Behaviour of bears in front of electrified barriers (cross or no cross into melon patch or fenced road corridor).

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Research Article

Influence of land use intensity on ecological corridors and wildlife crossings' effectiveness: comparison of 2 pilot areas in Austria

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Abstract

Human development and induced activities significantly affect the natural functioning of ecosystems and hence landscape connectivity. Ecological corridors are essential for maintaining structural as well as functional connectivity in cultural landscapes for wildlife, while providing interchange between core areas. In two pilot areas in the north-western and eastern part of Austria, ecological corridors were delineated using a geographic information system (GIS). The pilot areas are key to preserving ecological connectivity and are located along important international migration corridors (Bohemian Forest-Northern Alps corridor, Alpine-Carpathian corridor). Both areas are situated in highly human-altered and therefore dissected as well as fragmented landscapes. A one-year monitoring campaign using camera traps was carried out at selected locations along proposed ecological corridors in the cultural landscape and at wildlife crossings structures (WCSs) at intersections with road infrastructure. The monitoring was focused on mammals with a total of 18 species being observed. The most abundant species were roe deer, European hare and wild boar. European otter, European beaver, golden jackal and wildcat have only rarely been observed. Mammal species richness was positively correlated with the presence of vegetation cover and the coefficient of ecological stability (CES). The insights obtained can be used for recommendations and support in planning the planting of vegetation (use of grasslands, scattered and continuous woody vegetation, agroforestry systems) on the sites and in the vicinity of ecological corridors. The green bridges (wildlife overpasses) were used more frequently as well as by a larger number of mammal species compared to other studied WCSs showing characteristics that are less favourable for animals. The effectiveness of WCSs is mainly influenced by human activities, resulting in the recommendation to limit them on WCSs located along the routes of ecological corridors. We point out that actual wildlife migration corridors are likely to differ from designated data-driven ecological corridors generated by spatially explicit models, because these generally do not take into account all factors relating to the effectiveness of corridors. Our results suggest, that the application of the concept of functional connectivity is able to enhance the quality of ecological corridor designations, since usually they are based only on the concept of structural connectivity. For this reason, further studies are needed to help understanding factors and their specificities influencing the interplay between structural and functional connectivity of ecological corridors.

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Key words: Alpine-Carpathian corridor, Bohemian Forest-Northern Alps corridor, coefficient of ecological stability, daily activity, landscape connectivity, functional connectivity, habitat fragmentation, wildlife crossing structures, wildlife monitoring

Introduction

Landscapes are changing dramatically due to human influences (Gardner et al. 1993; Antrop 2004; Bennett and Saunders 2010). The current state of the landscape is the result of significant changes caused by urban and rural development, the expansion of linear infrastructure and traffic, agricultural intensification, changes in forest management, expansion of energy networks and renewable energy (Antrop 2004; Farinaci et al. 2014; Van Der Ree et al. 2015b; Plieninger et al. 2016). Around 80% of the Earth's land surface currently shows signs of human intervention (Ellis and Ramankutty 2008; Venter et al. 2016a, 2016b) resulting in loss of biodiversity (Butchart et al. 2010) and of the capacity and multifunctionality of ecosystem services (Corvalán et al. 2005). In the EU and UK, almost a third of the land (27%) is highly fragmented where habitats are on average less than 0.02 km^2 (EEA 2022).

Unscathed and coherent natural habitats are gradually being disintegrated by humans into smaller units or spatially disjunct patches (Bennett and Saunders 2010). Fragmentation results in species behavioural changes (Opdam et al. 1993; Tucker et al. 2018) connected with a number of serious problems such as habitat loss (Huxel and Hastings 1999; Brooks et al. 2002), landscape change (Antrop 2004; Leimu et al. 2010; Jarzyna et al. 2015), biodiversity loss and reduced fitness of wild animals due to genetic isolation and inbreeding (Ellegren et al. 1996; Hanski 2011; Lino et al. 2019) leading to wildlife population decline (Bender et al. 1998) or extinction (Andrén 1997; Fahrig 1997; Pardini et al. 2018; Wilkinson et al. 2018) and increased wildlife-vehicle collisions (Morelle et al. 2013; Vanlaar et al. 2019; Saint-Andrieux et al. 2020). Mitigation or so-called defragmentation measures are applied in practice to avoid or minimise the above-mentioned consequences. In order to ensure permanent permeability of the landscape for wildlife, ecological corridors or networks (Jongman et al. 2011; Gregory et al. 2021) and wildlife crossings structures (WCSs) (Ford et al. 2009; Smith et al. 2015) are planned, built and maintained. Ensuring connectivity is among the main contemporary challenges in the protection of nature and landscape (Bennett et al. 2006; Jongman et al. 2011; Keeley et al. 2018).

Ecological corridors are important elements in nature and landscape conservation. They are the backbone of green infrastructure necessary to maintain or restore connectivity, biodiversity and ecological functions in the landscape (Bennett and Mulongoy 2006; Zheng et al. 2019; Gregory et al. 2021). Ecological corridors are usually linear-shaped areas providing connections between native habitats, stepping stones, and interacting features in the landscape while enhancing the ability of wildlife and plants to move between larger core areas (Bennett and Mulongoy 2006; Damschen et al. 2006; Hilty et al. 2020; Gregory et al. 2021). Connecting the remaining habitat fragments plays an important role in maintaining gene flow and genetic diversity for both plants (Sork and Smouse 2006) and wildlife (Waits et al. 2015). WCSs are an important instrument, ensuring the connectivity of ecological corridors through high-density traffic infrastructure. Increasingly, integrated approaches of geographical information systems (GIS) and other software tools are being used to design and manage appropriate routes for ecological corridors including localization and parameterisation of WCSs. These spatially explicit methods use a number of different algorithms such as cost distance (least-cost path), circuit theory or Euclidean distances (McRae et al. 2008; Suppan and Frey-Roos 2014; Loro et al. 2015; Ribeiro et al. 2017; Zheng et al. 2019; Mardeni et al. 2023). Two different aspects of connectivity should be distinguished, both of which are prerequisites for the success of corridors: in contrast to structural connectivity, which considers the physical characteristics that support connected habitats, functional connectivity can be defined as the degree of specific responses of certain wildlife to elements in the landscape (Koen et al. 2014; Sedy et al. 2022).

The effectiveness of WCSs has already been investigated in many studies (Clevenger and Waltho 2003; Mata 2003; Simpson et al. 2016; Mysłajek et al. 2020). However, most of the studies focused on the effects of size and type of WCS, rather than the characteristics of the location of WCS. Moreover, the majority of existing studies are based only on the number of individuals crossing WCSs and do not account for the number of total approaches (Chambers and Bencini 2015). Denneboom et al. (2021) summarised the results of 77 studies regarding WCS efficiency in a systematic review and meta-analysis. They showed that viaducts are the most effective type of WCS for large mammals and that WCSs built specifically for wildlife are used significantly more than those used by humans in addition to wildlife.

The aim of this work was to conduct monitoring of the occurrence of mammals at designated sites (EEA and Jurečka 2023) along ecological corridors within the cultural landscape and WCSs at crucial intersections with road infrastructure in two pilot areas in Austria. Subsequently, the data of mammal presence and species richness were compared with the current state of the environment near the monitoring sites using the ecological stability coefficient (the ratio of relatively stable areas to unstable areas). The influence of distance to various landscape features (core areas, forested areas, watercourses, water bodies, motorway incl. expressway infrastructure, built-up areas) was also evaluated. The last objective of the work was to identify the permeability for mammals on the studied WCSs located along the routes of ecological corridors designed using an integrated approach employing geographic information systems combined with human activity and width of WCSs.

Methods

a) Ecological corridors: GIS-model

Ecological corridors designated for the Interreg Danube project SaveGREEN, which are located in two pilot areas in Upper Austria as well as at the border between Lower Austria and Burgenland, were used as starting point (Sedy et al. 2022; EEA and Jurečka 2023). These corridors were designed with the aid of geographic information systems. Factors such as land cover, altitude, slope, presence of watercourses, presence of wildlife crossing structures, buildings, road and railway networks were taken into account. Two input layers were modelled: i) the extent of core areas (including stepping stones), and ii) the surface resistance in terms of wildlife migration, both provided the basic framework for modelling the route of ecological corridors in the pilot areas (Plutzar and Sedy 2021a, 2021b). Core areas can be understood as areas that provide long-term suitable habitat for wildlife, whereas stepping stones are small patches of habitat in the landscape that provide rest and shelter during wildlife migration. Surface resistance refers to the degree of obstacles that wild animals have to overcome during their migration in a fragmented landscape. The ecological corridor model was created using habitat connectivity analysis and the tool Linkage Mapper (McRae and Kavanagh 2011). The first pilot area was located west of the provincial capital city of Linz in Upper Austria (Kobernausser Forest Pilot Area, hereafter KF). The KF landscape (approx. 5 000 km^2) presents itself as a hilly area divided by shallow, mostly unobstructed stream valleys and mainly covered by spruce forests. The second pilot area (Pöttsching Pilot Area, after the nearby municipality, hereafter PÖ) was located south of Vienna in the border area between Lower Austria and Burgenland. From a morphological viewpoint, PÖ (approx. 1 200 km^2) is largely located in a flat or undulating hill country at the edge of the Pannonian Plain. Both pilot areas are embedded in highly human-fragmented landscapes and at the same time are part of important international migration corridors for wildlife (Fig. 1). KF covers parts of an important migration corridor from southern Bohemia to the Central Alps (Bohemian Forest-Northern Alps corridor), while PÖ lies in the Alpine-Carpathian corridor (Birngruber et al. 2012; BMK and EAA 2023). Both areas represent critical sections or bottlenecks along these international corridors (Woess et al. 2002; Birngruber et al. 2012) and are essential in terms of maintaining connectivity in the landscape and permeability for wildlife on the local, supraregional and transnational level.

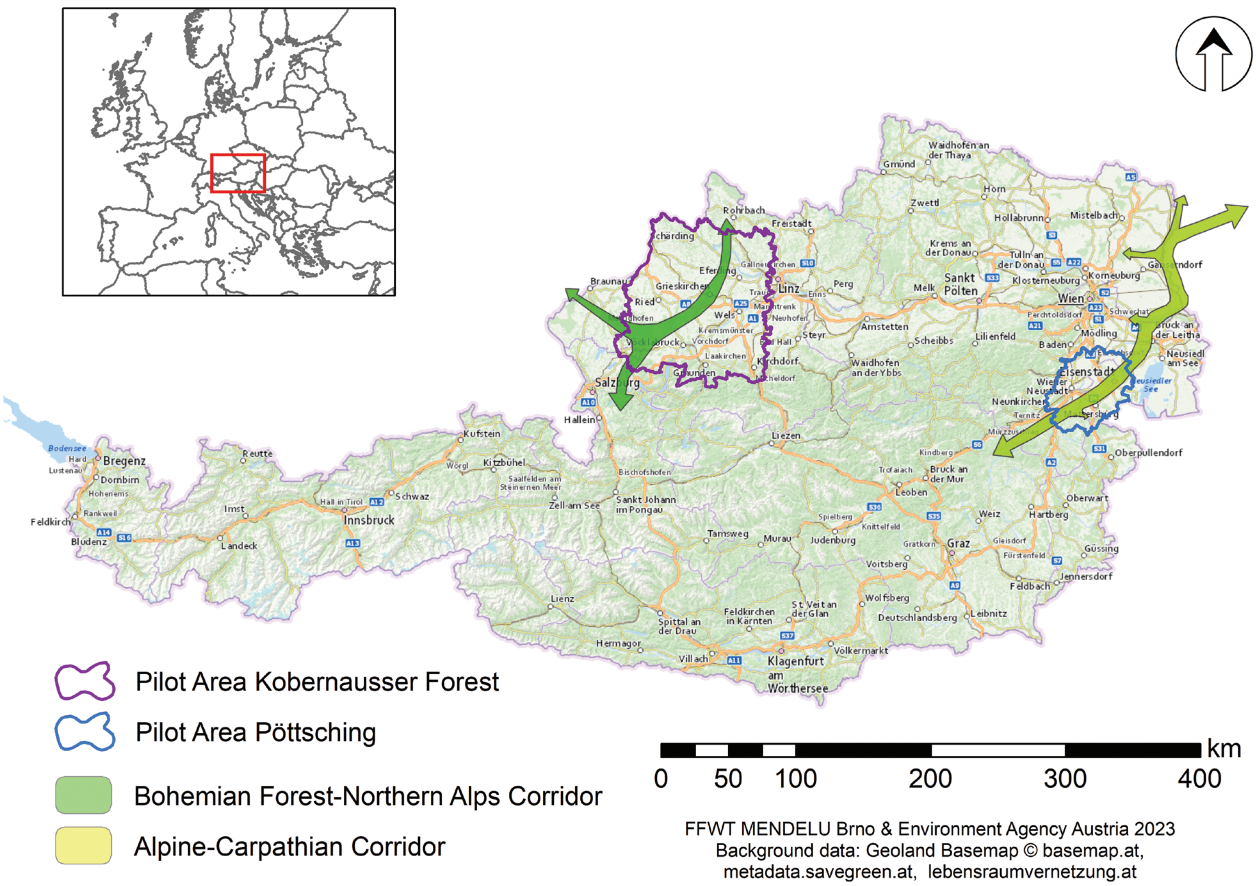

Figure 1. Location of the two pilot areas in Austria and indicative routes of important migration corridors with international connections.

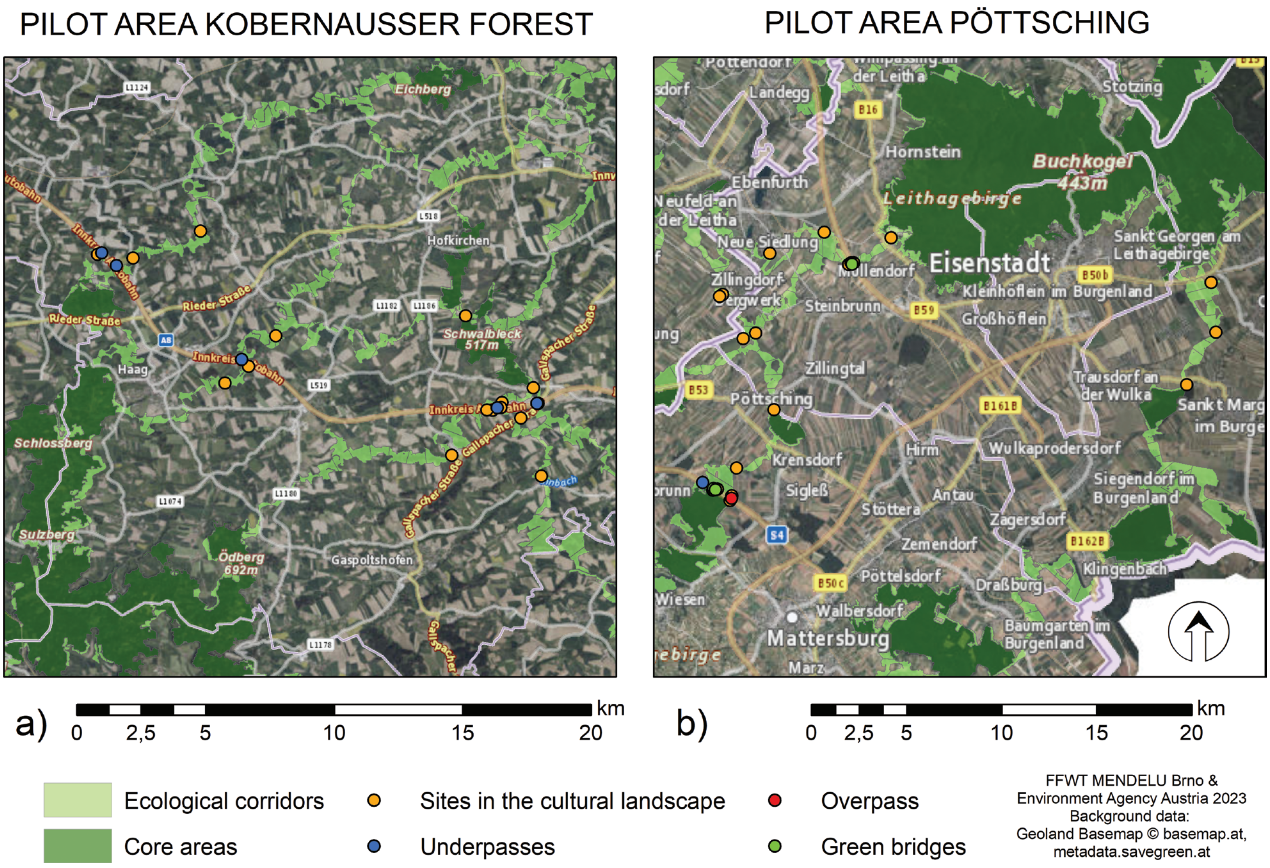

Figure 2. Location of sites with long-term monitoring by camera traps along ecological corridors and on wildlife crossing structures (EEA and Jurečka 2023).

b) Ecological corridors: monitoring sites

We selected a total of 49 sites along ecological corridors incl. WCSs for long-term monitoring, i.e. 21 sites at KF (Fig. 2a) and 28 sites at PÖ (Fig. 2b). The term “site/s” refers to the place and its immediate surroundings where the monitoring was conducted (Fig. 3). All sites were selected on the results of the GIS corridor modelling as part of the SaveGREEN project, the presence of wildlife tracks and routes, and also in coordination with, and in consent with, stakeholders (landowners, hunting associations, land users, mayors, etc.). Each site was provided with an information plate about the ongoing research and contact details. Nine wildlife crossing structures were monitored along the intersection of ecological corridors with motorways and expressways, including 5 underpasses in KF and 2 green bridges (GB), 1 underpass (U) and 1 grey overpass (O) in PÖ (Table 1). The underpasses were featured with asphalt roads and paved surfaces (may therefore be considered as grey underpasses). The grey overpass was equipped with asphalt roads complemented by guardrails and the green bridges were covered with natural surfaces and vegetation. One camera trap was used at each site. A larger number of camera traps were used at green bridges and the underpass due to the representative coverage of the object. Each green bridge was covered by 4 monitoring sites, one underpass (U5) by 2 monitoring sites. For further processing these sites were merged and thereby only one object representing the WCS was used in the analyses. The other 33 sites were located in the cultural landscape. For long-term monitoring, automatic camera traps “COOLIFE Wildlife Camera” were used. Monitoring took place from 16 December 2021 to 16 January 2023 for the KF and from 2 December 2021 to 28 January 2023 for the PÖ. Central European Time was used uniformly throughout the monitoring period. Camera traps were inspected every 4 to 8 weeks and data downloaded. By using ExifPro 2.1 software, the data was manually sorted by wildlife species or human activity (8 categories, i.e. pedestrians, pedestrians with dogs, cars, agricultural and forestry machinery, cyclists, motorcyclists, horse riders and others), abundance (number of individuals per record), site identification, time and date. For the subsequent objective evaluation of each site, the operating days of each camera traps were recorded. The monitoring methodology focussed specifically on mammals. Individual mammals that could not be determined directly to species were determined only at the genus level. Whereas mammals that remained unidentifiable were included in the special category "undetermined". Regarding the analysis of the relationship of the effect of human frequency on the number of mammal crossings in WCSs, data on human activities obtained from records for the majority of WCSs was used, however only for the underpass U5, traffic count data was used, i.e. 4,533 vehicles per day, annual average daily traffic of the year 2022 (DORIS 2023). A total of 56,717 mammal and human relevant records were obtained during the monitoring period, divided into 28,128 records from KF and 28,589 records from PÖ.

Table 1. Monitored wildlife crossing structures.

ID	Type WCS	Name	GPS coordinates (WGS84)	Road ID	Width of WCSs (m)	Pilot area
U1	Underpass	Straß	48°12'52.4"N, 13°37'31.6"E	A8	6	KF
U2	Underpass	Renhartsberg	48°12'40.6"N, 13°37'48.2"E	A8	5	KF
U3	Underpass	Rampersdorf	48°11'21.7"N, 13°40'26.0"E	A8	6	KF
U4	Underpass	Thalheim	48°10'41.3"N, 13°45'48.3"E	A8	8	KF
U5	Underpass	Niederetnisch	48°10'45.2"N, 13°46'38.7"E	A8	40	KF
U6	Underpass	Bad Sauerbrunn	47°46'44.8"N, 16°21'33.6"E	S4	70	PÖ
O	Overpass	Sigleß	47°46'26.1"N, 16°22'22.6"E	S4	7	PÖ
GB1	Green bridge	Pöttsching	47°46'37.3"N, 16°21'53.9"E	S4	80	PÖ
GB2	Green bridge	Müllendorf	47°50'55.4"N, 16°25'47.1"E	A3	50	PÖ

Note: The width of the WCSs was measured parallel to the intersecting road and was measured approximately using GIS (using orthophotos). WCSs were named after the nearest municipality.

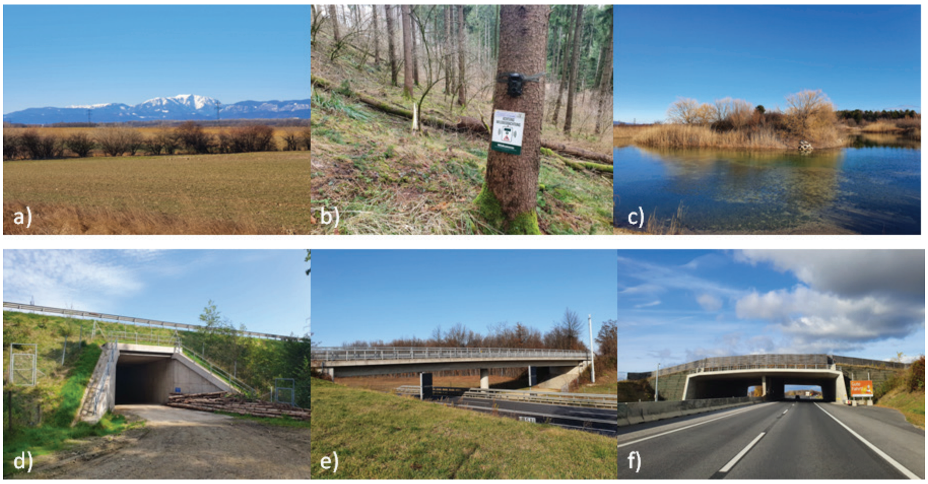

Figure 3. Monitoring sites along ecological corridors incl. WCSs: sites in the cultural landscape **a** farmland in PÖ **b** forest habitat in KF **c** standing water in PÖ **d** underpass (U1) of the A8 Innkreis motorway **e** grey overpass (O) over the S4 Mattersburger expressway **f** green bridge (GB2) near Müllendorf over the A3 Südost motorway (photos: Mořic Jurečka).

c) Ecological corridors: descriptive variables

For the spatial assessment and map output, the landcover layer EUNIS Biotoptypen Österreichs 2018 (EEA 2023) and additionally road infrastructure, watercourses, water bodies (data.gv.at 2023; geoland.at 2023), core areas and ecological corridors (EEA and Jurečka 2023) were used. Spatial data was processed with ArcMap 10.4.1. (ESRI 2015) using the ETRS 1989 LAEA coordinate system for reasons of suitability for the territory of Austria. To determine the degree of ecological stability and the degree of human disturbance in the vicinity of the monitoring sites, information on landcover (habitat types EUNIS 2018) in a circular buffer was used to evaluate the CES (Míchal 1982; Löw 1995; Chromčák et al. 2021). A diameter 1000 m was chosen with respect to the width of corridors of international importance and as recommended for international corridors in Upper Austria (Birngruber et al. 2012). The coefficient of ecological stability (CES) shows the ratio of relatively stable (S) to relatively unstable (UN) areas in a particular area, represented by the following expression:

$$CES = \frac{S}{UN} = \frac{X + F + E + G + C}{J + I}$$

To calculate the CES value via layer land cover (Table 2), the CES formula was used for each locality and the methodology of the Czech Statistical Office (Table 3) was used to interpret the results (Czech Statistical Office 2023). Euclidean distances were used to determine the proximity between monitoring sites and landscape features with potential impacts on mammals.

Table 2. Legend of relevant layers of EUNIS 2018 habitat types (EEA 2023) used for the analyses of CES.

Layer ID	Description of EUNIS habitat types
J	Constructed, industrial and other artificial habitats
I	Regularly or recently cultivated agricultural, horticultural and domestic habitats
X	Habitat complexes
F	Heathland, scrub and tundra
E	Grasslands and lands dominated by forbs, mosses or lichens
G	Woodland, forest and other wooded land
C	Inland surface waters

Table 3. Possible interpretation of the CES (Czech Statistical Office 2023).

CES value	Explanation
CES < 0.10	areas with maximum disturbance of natural structures
0.10 < CES < 0.30	areas with above-average use, with clear disturbance of natural structures
0.30 < CES < 1.00	areas intensively exploited, especially by large-scale agricultural production, weakening of autoregulatory processes in ecosystems
1.00 < CES < 3.00	areas with a broadly balanced landscape in which human influence is relatively consistent with preserved natural structures
CES > 3.00	areas with natural and close to nature landscapes with a significant predominance of ecologically stable structures and low intensity of human use of the landscape

d) Statistical analysis

In order to scrutinize the relationship between wildlife activities and the characteristics of the surroundings of the WCS, the recorded mammal data obtained from terrestrial monitoring at each site and the outputs from the spatial analysis were statistically compared using R software (R Core Team 2022). The processing was carried out using data on CES values, distances (abbreviated: DIST) from landscape features, the number of species at the sites and the average daily activity (hereafter ADA) of mammals at the site (average of total mammal records at the site to the number of operation days of the camera trap) and selected large mammal species (red deer, wild boar, roe deer). These were selected as representative species given their habitat requirements according to the proposed aspects of the ecological corridors (Plutzar and Sedy 2021a, 2021b) as well as potential risk and damage of wildlife-vehicle collisions. In treating the effect of distance from the motorways incl. expressways, only sites along ecological corridors outside of wildlife crossing structures were processed.

The software R (R Core Team 2022) was used to create scatterplots, correlation diagrams as well as other graphs and served for statistical testing. A linear regression (represented by a red line) has been fitted to the scatterplots. Spearman's correlation coefficient (r) was used to test and evaluate the relationship between the number of species and their activity in relation to landscape features, CES value, human activity and WCSs width. Mammals ADA with respect to the type of WCS was compared by applying the two-sample Wilcoxon rank sum test (Hollander and Wolfe 1973).

Results

a) Descriptive analysis: mammals

A total of 18 mammal species was recorded on the ecological corridors including WCSs during the monitoring period (Table 4). The highest number of species was found in the PÖ (16 species compared to 10 species in the KF). The most abundant mammal was roe deer (*Capreolus capreolus*) (51.24%), followed by European hare (*Lepus europaeus*) (20.51%) and wild boar (*Sus scrofa*) (8.39%). In contrast, the least abundant mammals registered were European beaver (*Castor fiber*), hedgehog (*Erinaceus* spp.), European rabbit (*Oryctolagus cuniculus*), European otter (*Lutra lutra*), European wildcat (*Felis silvestris*), least weasel (*Mustela nivalis*), European fallow deer (*Dama dama*), European mouflon (*Ovis aries musimon*), red deer (*Cervus elaphus*) and golden jackal (*Canis aureus*). European beaver, European muflon, European otter, European rabbit, European fallow deer, golden jackal, least weasel and red deer were recorded only in PÖ whereas wildcat and hedgehog were recorded exclusively in KF. The majority of records have been obtained on ecological corridor sites in the cultural landscape (61.05%) compared to WCSs (38.95%).

Table 4. Mammal presence in pilot areas on ecological corridors incl. WCSs.

Species	KF		PÖ	
	n	%	n	%
domestic cat (*Felis catus*)	1924	13.93	103	0.46
European badger (*Meles meles*)	64	0.46	213	0.95
European beaver (*Castor fiber*)	–	–	49	0.22
European hare (*Lepus europaeus*)	3180	23.02	4253	18.96
European mouflon (*Ovis aries musimon*)	–	–	203	0.91
European otter (*Lutra lutra*)	–	–	6	0.03
European rabbit (*Oryctolagus cuniculus*)	–	–	13	0.06
European wildcat (*Felis silvestris*)	3	0.02	–	–
European fallow deer (*Dama dama*)	–	–	2	0.01
golden jackal (*Canis aureus*)	–	–	1	0.01
hedgehog (*Erinaceus* sp.)	40	0.29	–	–
least weasel (*Mustela nivalis*)	–	–	3	0.01
marten (*Martes* sp.)	549	3.97	822	3.66
red deer (*Cervus elaphus*)	–	–	699	3.12
red fox (*Vulpes vulpes*)	390	2.82	1566	6.98
red squirrel (*Sciurus vulgaris*)	155	1.12	208	0.93
roe deer (*Capreolus capreolus*)	7487	54.20	11085	49.42
wild boar (*Sus scrofa*)	3	0.02	3039	13.55
undetermined	18	0.13	165	0.74

b) Descriptive analysis: humans

The number of records of human activities in KF was almost twice as large compared to PÖ (Table 5). Overall, cars were more than half as represented (53.72%), followed by human activity categories such as pedestrians (17.54%), agricultural and forestry machinery (11.66%), cyclists (7.84%), pedestrians with dogs (5.51%), motorcyclists (1.62%), other categories (1.28%) and horse riders (0.84%).

Table 5. Human presence in pilot areas along ecological corridors incl. WCSs.

Human activity	KF		PÖ	
	n	%	n	%
agricultural and forestry machinery	1624	10.17	1302	14.27
cars	10111	63.34	3365	36.88
cyclists	1201	7.52	765	8.38
horse riders	8	0.05	202	2.21
motorcyclists	235	1.47	171	1.87
others (excavators, trucks, etc.)	258	1.62	62	0.68
pedestrians	1946	12.19	2454	26.89
pedestrians with dogs	579	3.63	804	8.81

c) Patterns of mammal and human activity

Mammal activity was recorded mainly during the night hours. Throughout the year increased levels of activity of mammals was observed during dawn and dusk (Fig. 4). The highest frequency of activity of mammals was recorded in spring months. Human activity was recorded mainly during day for the entire year (Fig. 5). Human activity was predominant in the morning and afternoon hours. However, a decrease in human activity was recorded, especially in the midday hours throughout the year.

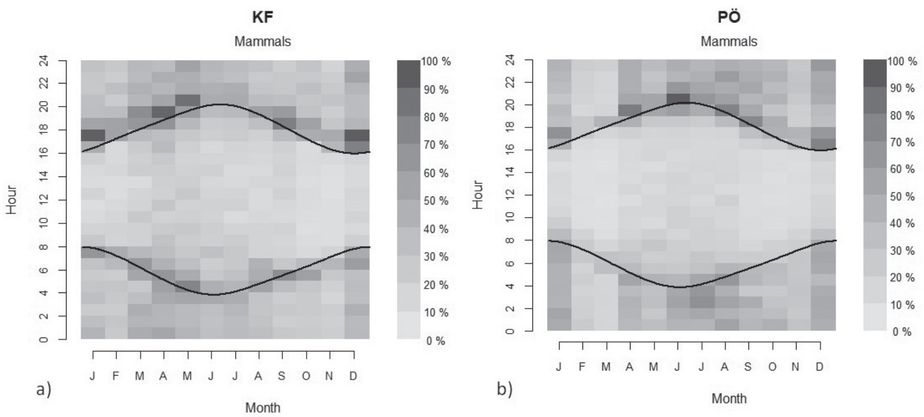

Figure 4. Annual and daily time distribution of mammal activity in KF (**a**) and PÖ (**b**). The solid black line represents the sunrise and sunset times during the year.

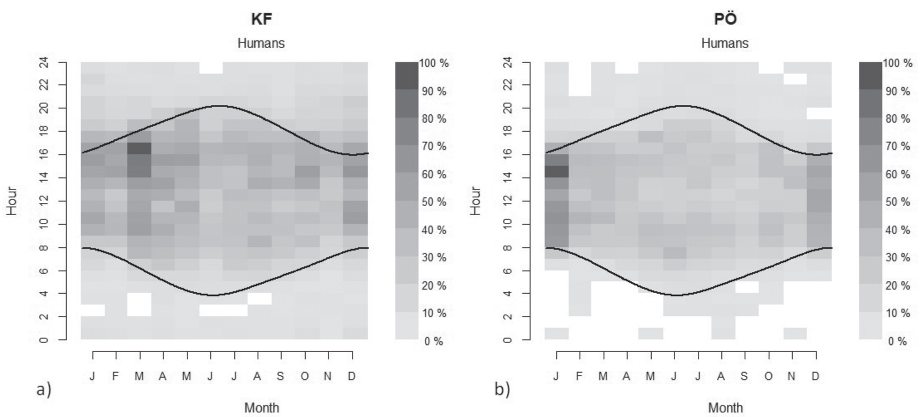

Figure 5. Annual and daily time distribution of human activity in KF (**a**) and PÖ (**b**). The solid black line represents the sunrise and sunset times during the year.

d) Coefficient of ecological stability

Almost 80% of all monitored sites on ecological corridors showed a CES value less than 1, indicating that these areas are characterised by disturbed natural structures and intensive human use (Fig. 6).

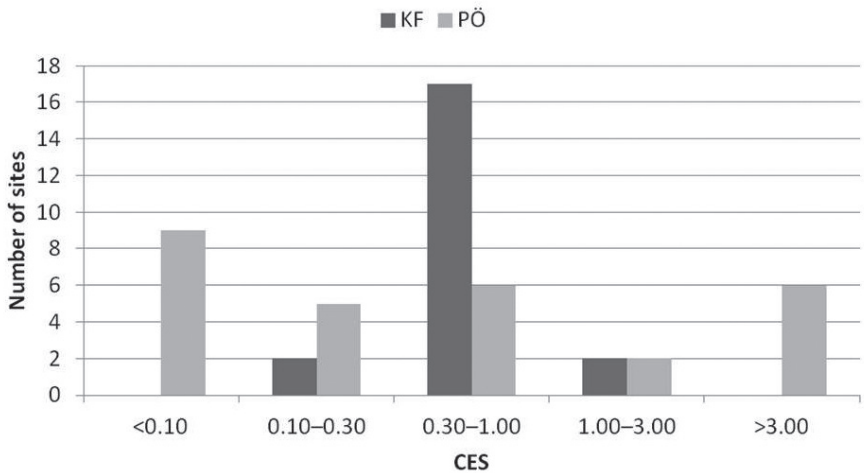

Figure 6. Coefficient of Ecological Stability (CES) values by pilot area monitoring sites.

A positive correlation (r = 0.29, p-value = 0.0471) was found between the number of species recorded and the CES value (Fig. 7a). Furthermore, a positive correlation (r = 0.26, p-value = 0.0689) was also observed with the ADA of mammals and the CES value (Fig. 7b).

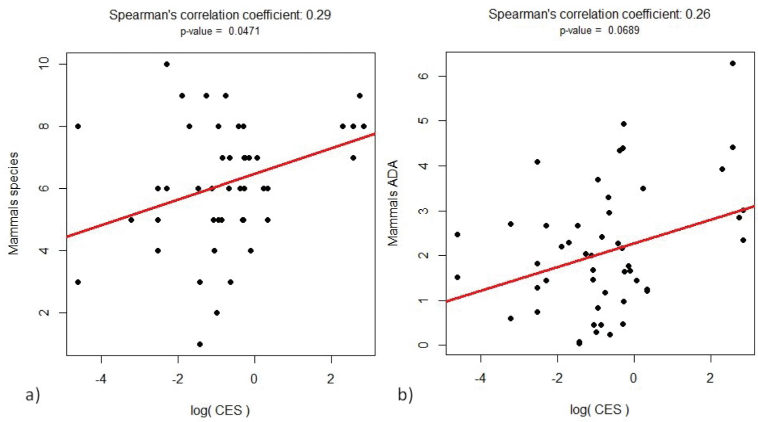

Figure 7. a Relationship between the number of species and **b** average daily activity (ADA) of mammals, and the logarithm of the ecological stability coefficient (CES), linear regression is shown in red.

e) Landscape features

The relationship of mammal presence as well as mammal activity and selected landscape features is presented in a correlation matrix (Fig. 8). Strong correlations were found between some related environmental parameters, for example between distance from built-up areas and distance from motorways. The strongest significant positive correlation with CES value was observed for ADA of red deer. The strong significant negative correlation with the CES value was observed for distance from core areas and also distance from forests.

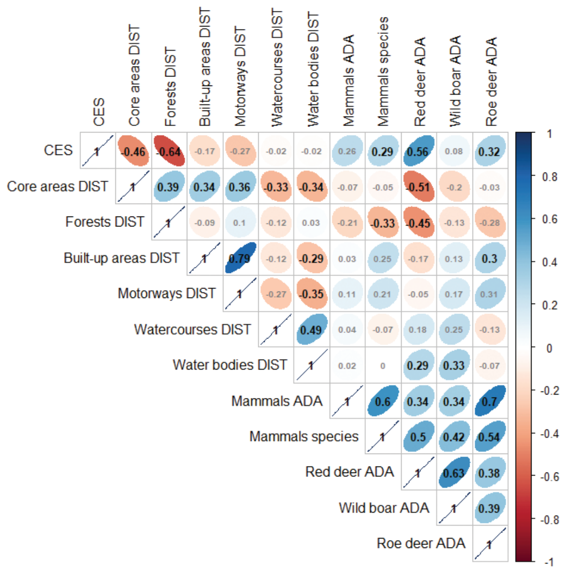

Figure 8. Multi-factor correlation matrix (CES, DIST: distance, ADA: average daily activity). Significant correlation coefficients (at significance level of 5%) are displayed in black while non-significant coefficients are presented in grey. The right side of the figure shows a scale for the correlation coefficient r indicating the colour codes used.

The average daily activity of red deer was influenced by environmental parameters. Significant negative correlations with red deer ADA were found for distance from core areas and forests, significant positive correlations were found for CES and distance from water bodies. The average wild boar daily activity was significantly positively correlated with distance from water bodies. The wild boar daily activity significantly positively correlated with red deer activity and activity of all mammals. The average roe deer daily activity was significantly positively related to CES and distance from built-up areas. The daily activity of roe deer was significantly positively correlated with red deer and wild boar activities, as well as with activity of all mammals combined.

f) Wildlife crossings structures

Altogether, 11 species were identified at the investigated WCSs (Table 6). The highest number of species was recorded on the green bridges (GB1, GB2) and at the U1 underpass. Average daily mammal activity at the green bridges is about 12.45 crossings per day (n = 2), approx. 3 times more than at the surveyed overpass (n = 1) and approx. 6 times more than the average daily mammal activity at all surveyed underpasses (n = 6). The highest number of species and the highest level of daily activity was recorded on the green bridge GB1 near Pöttsching, whereas the lowest number of species and the lowest daily activity was recorded on the underpass U5 near the residential area Niederetnisch. Only domestic cat

Table 6. Mammal and human presence in WCSs.

	U1		U2		U3		U4		U5		U6		O		GB1		GB2	
	n	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*	*n*	*ADA*
domestic cat (*Felis catus*)	178	0.48	142	0.36	838	2.55	395	1.04	21	0.07	10	0.02	18	0.26	14	0.04	5	0.01
European badger (*Meles meles*)	–	–	–	–	–	–	–	–	–	–	–	–	2	0.03	37	0.10	75	0.25
European hare (*Lepus europaeus*)	143	0.38	154	0.39	457	1.39	230	0.60	2	0.01	67	0.16	32	0.47	526	1.46	1640	4.38
golden jackal (*Canis aureus*)	–	–	–	–	–	–	–	–	–	–	–	–	1	0.01	–	–	–	–
hedgehog (*Erinaceus* spp.)	7	0.02	–	–	–	–	24	0.06	–	–	–	–	–	–	–	–	–	–
marten (*Martes* spp.)	34	0.09	8	0.02	66	0.20	32	0.08	9	0.05	–	–	222	3.26	53	0.15	37	0.10
red deer (*Cervus elaphus*)	–	–	–	–	–	–	–	–	–	–	4	0.01	–	–	515	1.36	–	–
red fox (*Vulpes vulpes*)	5	0.01	259	0.66	18	0.05	17	0.04	–	–	20	0.05	20	0.29	209	0.57	323	0.85
red squirrel (*Sciurus vulgaris*)	1	0.00	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
roe deer (*Capreolus capreolus*)	527	1.42	290	0.73	238	0.73	69	0.18	–	–	382	0.93	–	–	3304	9.07	800	2.29
wild boar (*Sus scrofa*)	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1499	3.97	4	0.01
undetermined	7	0.02	2	0.01	–	–	–	–	–	–	13	0.03	–	–	94	0.25	16	0.04
Total mammal activity	**902**	**2.42**	**855**	**2.16**	**1617**	**4.93**	**767**	**2.01**	**32**	**0.13**	**496**	**1.21**	**295**	**4.34**	**6251**	**16.97**	**2900**	**7.94**
Species richness	7	–	5	–	5	–	6	–	3	–	5	–	6	–	8	–	7	–
agricultural and forestry machinery	47	0.13	247	0.63	683	2.08	456	1.20	–	–	952	2.32	189	2.78	–	–	1	0.00
cars	373	1.00	169	0.43	1126	3.43	8259	21.68	*	4533	3046	7.43	312	4.59	–	–	–	–
cyclists	52	0.14	–	–	10	0.03	1117	2.93	–	–	690	1.68	37	0.54	–	–	34	0.10
horse riders	1	0.00	–	–	–	–	2	0.01	–	–	173	0.42	13	0.19	–	–	16	0.05
motorcyclists	6	0.02	–	–	22	0.07	176	0.46	–	–	151	0.37	1	0.01	–	–	3	0.01
others (excavators, trucks, etc.)	75	0.20	75	0.19	7	0.02	101	0.27	–	–	47	0.11	2	0.03	–	–	1	0.00
pedestrians	213	0.57	164	0.42	98	0.30	930	2.44	2	0.01	1576	3.84	276	4.06	135	0.39	183	0.49
pedestrians with dogs	8	0.02	18	0.05	121	0.37	348	0.91	–	–	535	1.30	62	0.91	18	0.05	54	0.16
Total human activity	**775**	**2.08**	**673**	**1.70**	**2067**	**6.30**	**11389**	**29.89**	**2**	**4533**	**7170**	**17.49**	**892**	**13.12**	**153**	**0.44**	**292**	**0.82**

Note: ADA – average daily activity in this regard refers to the average daily crossings of a species, * – traffic count data was used (DORIS 2023).

and hare were recorded on all WCSs surveyed. Smaller mammals such as squirrel and hedgehog were only recorded at the underpasses. A rare record of a golden jackal was observed on the grey overpass in the PÖ. Large mammals such as red deer and wild boar clearly preferred, in terms of mammals ADA, the green bridges compared to the underpasses and grey overpasses examined (p-value = 0.0278, one-sided Wilcoxon rank sum test). Red deer were occasionally recorded at the nearby underpass, however the majority of crossings were recorded on the green bridge GB1 (99.23% of all records). The highest average human activity per day was observed at the U4, U5 and U6 underpasses, while the lowest mean values were recorded at green bridges GB1 and GB2.

A positive correlation was noted between the width of WCSs and the number of species as well as the ADA of mammals; no relationship was observed for the width of WCSs and the average daily human activity (Fig. 9). These three correlation coefficients were statistically insignificant. A significant negative correlation was found between the average daily human activity on WCSs and (i) the number of species ($r = -0.68$, p-value = 0.0422), and (ii) the ADA of mammals ($r = -0.83$, p-value = 0.0083). Green bridges displayed the highest wildlife activity and the lowest human activity compared to the other types of WCSs (Table 6).

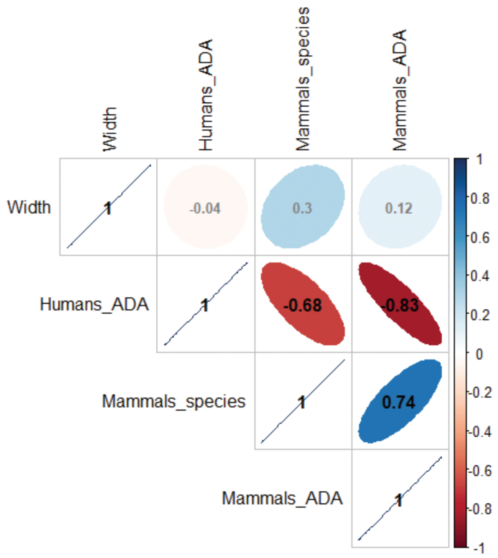

Figure 9. Correlation matrix of selected factors influencing the effectiveness of WCSs. Significant correlation coefficients (on the significance level of 5%) are displayed in black while non-significant coefficients are presented in grey. The right side of the figure shows a scale for the correlation coefficient r with the colour codes used.

Discussion

a) Wildlife monitoring at sites on ecological corridors and time variation

Maintaining structural and functional connectivity is crucial for the long-term sustainability and viability of wildlife populations as well as for safeguarding ecosystem functions in human-altered landscapes. Ecological corridors are serving the goal of maintaining connectivity in the landscape. The selected monitoring locations were chosen based on the given requirements and limitations, primarily caused by the willingness of landowners and land users, as well as by the corridor routes defined initially. To optimise coverage, it would be advisable to monitor the ecological corridors even more comprehensively using additional monitoring sites, in particular to compare the sites with regard to the occurrence of wildlife in the core areas.

The highest species richness was recorded in the pilot area Pöttsching (PÖ) compared to the pilot area Kobernausser forest (KF). The abundant species richness in the PÖ may be related to local conditions at the interface of three different biogeographical regions, i.e. the Alpine, Continental and Pannonian regions (EEA 2017) and due to its location on the Alpine-Carpathian corridor (BMK and EAA 2023). It is important to emphasise that not all species of mammals were recorded consistently at every location on the ecological corridors incl. wildlife crossings structures (WCSs). This may be related to the current degree of anthropogenic fragmentation of habitats in the pilot areas, which may be particularly problematic for sensitive species and large mammals such as red deer, wild boar or large carnivores. Motorways represent a significant genetic barrier for red deer, which, in contrast, does not apply to wild boar (Frantz et al. 2012). For example, red deer was recorded only in PÖ while wild boar was recorded in both pilot areas. Red deer were found near the core areas and forests, e.g. near and south of the GB1 green bridge (Rosalia Mountains), north of the municipality of Müllendorf (Leitha Mountains) or in the vicinity of the municipality of Sankt Margarethen in Burgenland. However, they could practically not be found in locations along the ecological corridors between these core habitats. Only in the vicinity of the core areas (Rosalia, Leitha Mountains) European mouflon and fallow deer were recorded as well, while no evidence was recorded in between those areas. In contrast, wild boar was recorded at almost all of the monitored sites in PÖ, but compared to KF, it was recorded at only two sites north of the A8 motorway and did not cross any monitored WCSs. In this context, it must be taken into account that ungulate populations and associated occurrence may be significantly influenced by hunting management. Large carnivores such as the grey wolf and the Eurasian lynx were not recorded as part of our monitoring. This may be due to the fact that the investigated pilot areas are already significantly fragmented and disturbed by humans, which represents a major problem for large mammals, causing them to seek other, less disturbed routes. Although there were occasional reports of observations of grey wolves and lynx in the vicinity of the pilot areas, they were not observed along the designated ecological corridors as part of this work (Birngruber et al. 2012; Kora 2023; Marucco et al. 2023). This is in line with Ripari et al. (2022) suggesting that human disturbance is a limiting factor for habitat selection of large carnivores in continental Europe. Human disturbance may not be the main problem - in the case of wolves, for example, genetic studies confirm their migrations of several hundred kilometres across densely populated parts of Europe (Andersen et al. 2015; Hindrikson et al. 2017). Another factor that noticeably affects the spread and recovery of wildlife populations, especially large carnivores, which must be taken into account, is hunting and illegal killing (Kaczensky et al. 2011). The known spread of the golden jackal in Europe (Spassov and Acosta-Pankov 2019; Frangini et al. 2022) was also substantiated by our study with one record on a grey overpass (O) near the municipality of Sigleß. It is unclear whether it was a local individual, e.g. from the area of Lake Neusiedl, where the first breeding of the golden jackal in Austria was described (Herzig-Straschil 2007), or a migrating individual from the Pannonian Plain or South-Eastern

Europe (Lanszki et al. 2006; Hatlauf et al. 2017; Spassov and Acosta-Pankov 2019). Furthermore, a rare occurrence of European wildcat was observed in the KF pilot area (near the municipality of Haag am Hausruck), which would be consistent with relevant studies (Friembichler and Slotta-Bachmayr 2013; Slotta-Bachmayr et al. 2017), which indicates a relatively suitable habitat and refers to historical records of occurrence in this area. In order to objectively verify the presence of European wildcat, genetic analysis would be necessary, to confirm or completely exclude whether it was an individual of a domestic cat or a hybrid (Pierpaoli et al. 2003; Slotta-Bachmayr et al. 2017). In the Wachau region (east-oriented from KF), a small breeding population of European wildcat was recently confirmed by genetic analysis (Gerngross et al. 2021), which supports the assumption of migrating individuals. Furthermore, semi-aquatic species such as European otter and European beaver were recorded only in PÖ at locations near water courses (near the municipality of Oslip). The European beaver probably spread from the Danube River or the Pannonian Plain, and is expected to increase in population size (Halley et al. 2021). Although the European otter was documented in almost the entire territory of Austria (Kranz and Poledník 2020), in our case only isolated occurrence records were registered, which may indicate a fragmented distribution. Moreover semi-aquatic species are difficult to monitor due to their specific habitat conditions (Mata et al. 2005) and could therefore be underrepresented. Species such as roe deer, brown hare, marten, European badger, domestic cats, red fox and squirrel, which are usually common in human-modified landscapes, were also abundantly recorded. For roe deer, an increase in population size is generally observed, which refers to its good adaptation to human-dominated landscapes and is also consistent with its high ranking among the most frequent victims of wildlife-vehicle collisions (Ignatavičius et al. 2020; Bíl et al. 2023). Some species of particularly small and medium-sized mammals may not have been recorded at every site due to the technical limitations of camera traps (Jumeau et al. 2017). The higher number of domestic cats recorded in KF should be emphasised, which is probably related to human presence and scattered built-up areas in the landscape.

Most mammal records were registered at night with a significant increase in activity around dawn and dusk. This general trend is consistent with a number of studies and corresponds to the high probability of collision between wild animals and vehicles at dawn and dusk (Morelle et al. 2013; Krukowicz et al. 2022). The peak activity of mammals during the year was recorded in spring, followed by summer and during the transition from autumn to winter. This is also supported by trends in wildlife-vehicle collisions recorded in other studies (Krukowicz et al. 2022; Bíl et al. 2023). Human activity was recorded mainly during the day and characterised by two peaks (in the morning and in the afternoon), which is also supported by various studies (Reilly et al. 2017; Lewis et al. 2021). Increased human activity in the afternoon is also reflected in human-caused traffic accidents (Krukowicz et al. 2022). The peak in human activity was recorded in spring and subsequently decreased, which is probably related to the increase in the frequency of agricultural management measures, or other activities conducted in the landscape. The categories of human activity were represented relatively balanced across the pilot areas. However, KF was clearly dominated by activities associated with the use of cars, which may have been influenced by the number of underpasses in KF.

b) The influence of the coefficient of ecological stability, vegetation cover and landscape features

In our study the ecological stability coefficient (CES), which is used to express ecological stability under human influence at regional scale (Chromčák et al. 2021) illustrates the characteristics of the surrounding area of the monitoring sites and its impact on mammals occurrence on ecological corridors. Furthermore, additional information of important landscape elements that potentially have influence on the occurrence of mammals were considered. The quality of the results of the GIS-based calculations of the CES values and the distances from landscape elements was probably influenced by the quality of the background data used, especially regarding spatial accuracy and actuality. Data used as the basis for the calculations of the CES relate to 2018 while field work was carried out from 2021 to 2023. This discrepancy may have an influence on the results.

The results suggest that ecological corridors can fulfil their function and ensure the movement of mammals (Table 4) even in human-modified landscapes such as those found in the regions (Fig. 6). CES emphasises areas that are stable in an ecological context such as grasslands, shrubs, scattered vegetation, woodlands, forests and habitat complexes. The results indicate that various types of vegetation and habitat structures can noticeably support the activity and migration of species, and in the case of mammals also lead to increased species richness. This is supported by the negative relationship between the number of species and the activity of mammals, especially when considering large mammals such as red deer, and the distance from core areas and forest complexes. These results emphasise the importance of ecological restoration of such landscapes, for example through enrichment with landscape structures along ecological corridors. Especially in the context of rural agricultural landscapes, the implementation of the concept of green infrastructure (GI) as a strategically planned network of natural and semi-natural areas could significantly improve the permeability of featureless landscapes between core areas. Studies therefore argue for the development of customised local GI maps to highlight local requirements and options for such GI and to provide decision support for investment in GI, since the visualisation of priority conservation areas in a spatially explicit manner could support decision-makers to optimally allocate limited resources for ecosystem conservation and restoration (Danzinger et al. 2021). Ecological corridors would benefit from different zones of protection with customised degrees of applied management. Therefore applied management such as using zones of continuous canopy vegetation together with a buffer zone of scattered vegetation with grass cover, is highly recommended. The use of agroforestry systems, i.e. the integration of trees and shrubs into farming practices, and the restoration of elements of GI in the vicinity of ecological corridors would also be suitable for promoting biodiversity and wildlife connectivity (Jose 2012; Dondina et al. 2019; Udawatta et al. 2019). Moreover, the use of additional GI elements such as hedges and small woody features can increase ecological stability in the landscape while combining productive and protective functions for agricultural landscapes. Depending on the particular species, each measure can naturally entail a number of restrictions, which makes cross-species consideration essential for the planning of implementation measures. Although this work has not scrutinized the width of ecological corridors, which is one of the important factors influencing the distribution of species, the use of the guiding principle “the bigger the better” can be recommended (Bond 2003; Samways et al. 2010; Ford et al. 2020).

Furthermore, there was a positive association between the number of species and increasing distance from motorways and built-up areas. This suggests that human presence and associated disturbance affects the distribution of species in the landscape (Barrueto et al. 2014; Dertien et al. 2021). This finding may indicate the influence of the barrier effect (Lodé 2000; Jacobson et al. 2016; Seiler and Bhardwaj 2020). A lower correlation value was recorded for species activities, which is probably related to the disproportion of sites near anthropogenic features or the results may have been biased by synanthropic animals (such as the domestic cat and marten). A positive correlation was observed for roe deer and less positive correlation for wild boar, which is probably related to the ecology of these species. A slightly negative correlation was recorded for red deer, which is certainly influenced by the disproportion of record occurrence and the small sample size. There was almost no relationship between water features (watercourses, water bodies) and species or their average daily activity. This is probably due to the relatively small sample size, which influenced the result regarding the weak dependence on water features for roe deer and the opposite trend for red deer and wild boar.

c) Wildlife crossings structures

In the study, 9 different wildlife crossing structures (WCSs) were monitored, which support the crossing of high-traffic transport infrastructure by the routes of the ecological corridors. The distribution of the number of WCSs types was not ideal in terms of variables and for the subsequent statistical processing. This distribution is probably the reason why no statistically significant correlation was found between the width of the WCSs and efficiency in terms of the number of species crossing and their average daily activity. The importance of not only the width but also other parameters (length, height, openness, slopes) of the WCSs is supported by a number of studies (Van Wieren and Worm 2001; Grilo et al. 2008; Mata et al. 2008; Mysłajek et al. 2020). However, it is also important to consider the shape of the object, material, subsoil, location and many other biotic or abiotic factors (Denneboom et al. 2021; Brennan et al. 2022). The specific demands and responses of each species to the width factor also need to be taken into account (Brennan et al. 2022). Our study identified 11 species on WCSs compared to 18 species on sites in the surrounding cultural landscape. The technical aspects of WCSs (design, size, location and densities of WCSs) may vary depending on the specific location and the particular animal species (small, medium or large fauna). Small fauna species (e.g. reptiles, amphibians, flightless insects) often constitute habitat specialists and therefore the design details and equipment of WCS should always be adapted with respect to the their requirements. For this reason as well as to ensure the maximum efficiency of the WCS for all species and to maintain functional connectivity in the landscape, it is necessary to take into account a number of existing practical guidelines and recommendations for the design and the maintenance of WCSs (Iuell et al. 2003; Van Der Ree et al. 2015a; Hlaváč et al. 2019; Reck et al. 2019, 2023a).

There was a statistically significant negative correlation between ADA of humans and ADA of mammals, including the number of species on WCSs. This indicates that human presence and associated human disturbance in the surrounding environment significantly influences mammal movement across WCSs. This is supported by a number of other studies showing the negative impact of humans on wildlife (Barrueto et al. 2014; Denneboom et al. 2021; Dertien et al. 2021). For this reason, it may be recommended to completely avoid or minimise human activities in and around important WCSs and along ecological corridor routes to ensure better efficiency.

Green bridges (wildlife overpasses) showed better efficiency compared to underpasses or grey overpasses, either in terms of daily crossing rate or total number of observed species. The effectiveness of WCSs in general is influenced by a number of parameters and factors (Mysłajek et al. 2020; Brennan et al. 2022) that have been mentioned above, however, in the case of green bridges the presence of vegetation cover and the possibility of crossing over road infrastructure is an unparalleled advantage in comparison to ordinary bridges (grey overpasses) or underpasses (grey underpasses). The distance to vegetation cover is one of the important factors affecting the passage of large mammals (Clevenger and Waltho 2005). In our case, the effectiveness of green bridges was influenced by lower levels of human activity compared to other types of WCSs. Although other studies show better effectiveness of wildlife overpasses compared to underpasses (Simpson et al. 2016; Mysłajek et al. 2020), there remains a need for all WCSs to meet the goals of mitigating the impacts of habitat fragmentation by restoring connectivity and reducing the risk of wildlife-vehicle collisions to ensure better traffic safety (Olsson et al. 2008; Smith et al. 2015; Simpson et al. 2016). Different species are known to use and have various responses to different types of WCSs (Clevenger and Waltho 2005; Mata et al. 2005), which should be considered when designing WCSs for target species in specific locations (Smith et al. 2015). The results show that green bridges were significantly preferred by large mammals such as red deer and wild boar but also in the case of roe deer compared to other types of WCSs. This result is supported by other studies that show a preference for wildlife overpasses by large ungulates and large carnivores compared to underpasses (Clevenger and Waltho 2005; Kusak et al. 2009; Simpson et al. 2016; Mysłajek et al. 2020). A rare record of a golden jackal was observed on a grey overpass. Surprisingly, the European badger was recorded only on the overpasses, although it was also found in the surroundings along the ecological corridors. The average daily crossing rate for European badger, red fox, marten and European hare was higher on overpasses compared to underpasses. Mysłajek et al. (2020) showed similar results for European badger and red fox, but the opposite in the case of the European hare and marten, which preferred underpasses.

d) Needs for planning and protection of ecological corridors in the landscape

The identified routes of ecological corridors using the integrated GIS-approach may not coincide with actual wildlife migration routes because model results are affected by the accuracy, updating of the input layers and the assumptions made on animal behaviour regarding the surface resistance. Furthermore, it depends primarily on the state of the landscape at a certain point in time, which in human-modified landscapes changes considerably over time, as well as on a number of other factors. For example, "least cost path" analyses should not be used for management in landscapes without knowledge of actual migration route data and potential risks of movement across the landscape (Fahrig 2007). Based on the above, frequent updating of input data, like migration routes and species occurrences is recommended for modelling ecological corridors, as well as the precise coordination and targeting of policy, legislative and spatial planning tools to protect ecological corridors.

The need for defragmentation of the European landscapes is currently receiving considerable attention, which has led, i.e. to the development of the European Defragmentation Map (EDM), which provides an overview of the ecological core areas and the connecting ecological corridors within and between member States. So far, this map integrates data for 17 European countries and 2 transnational areas (Böttcher et al. 2022). To reduce barrier effects from the Trans-European Transport Network (TEN-T) the EDM allows to estimate the extent of current and future fragmentation and indicates priority sections for implementing mitigation measures such as the construction of wildlife crossings to restore and reconnect habitats (Reck et al. 2023b). Our results support these efforts by contributing in-situ field data on two important trans-European migration corridors, namely the Bohemian Forest-Northern Alps corridor and the Alpine-Carpathian corridor, to this Europe-wide initiative.

Conclusions

Our study indicates considerable diversity and activity of mammal species as well as aspects of functional connectivity on ecological corridors in two pilot areas in Austria. Applying the ecological stability coefficient (CES), the influence of land use intensity and the related importance of the presence of vegetation cover was shown – the number of species recorded and their average daily activity increased with the CES value. Species richness increases with greater distance from built-up areas or infrastructure. The green bridges (wildlife overpasses) achieve the highest efficiency compared to other WCSs covered, but this difference in efficiency is influenced by the parameters of the individual WCSs. The present study also underlines the strong influence of human activity in the vicinity of WCSs on species richness and mammal activity.

Green bridges have proven to be an effective type of WCS that significantly supports crossing for multiple species. However, in planning and design (not only in the pilot areas in Austria) the long-term provision of comprehensive connectivity to both ecological corridor routes and important landscape features should not be neglected.

The issue of habitat fragmentation and landscape change is currently gaining in importance, due to its relevance for biodiversity loss. Human infrastructure and other associated obstacles pose essential problems and challenges for many animals. The examined ecological corridors in Austria indicate that, in addition to structural connectivity, the quality of functional connectivity is also of crucial importance, especially with regard to sensitive species such as large mammals, as insufficient functional connectivity results in reduced permeability. Last but not least, we emphasise that the issue of landscape connectivity is becoming increasingly important and therefore further studies are necessary, taking into account global, regional and local factors.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

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Data availability

All of the data that support the findings of this study are available in the main text.

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Research Article

Experimental study on improving the utilization rate of underpasses of bundled linear infrastructure on Tibetan Plateau

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Abstract

Wildlife crossing structures (WCSs) are an important measure to protect biodiversity and reduce human-wildlife conflict, especially for bundled linear infrastructure. The aim of this study was to evaluate two "management and behavioral" factors (salt blocks and feces) in relation to two "structural factors" (underpasses' dimension and distance of bundled linear infrastructure) along Qinghai-Tibet bundled linear infrastructure (Qinghai-Tibet railway alignment runs parallel to the Qinghai-Tibet highway) and Gonghe-Yushu bundled linear infrastructure (Gonghe-Yushu expressway is parallel to the Gonghe-Yushu highway) using infrared cameras. Eight underpasses were monitored in the Qinghai-Tibet railway and six in the Gonghe-Yushu expressway, with half of the induced experimental group and half of the control group in each area. The monitoring shows that the Qinghai-Tibet railway area has richer species diversity than the Gonghe-Yushu expressway area. Salt block and feces induction experiments showed that the relative abundance index (RAI) of the experimental and control groups did not reveal significant differences in both areas. In addition, we found that the wider the width of the underpasses, the higher the utilization rate of kiang (*Equus kiang*) and wolly hare (*Lepus oiostolus*). And the distance from the adjacent linear infrastructure was positively correlated with the frequency of wolly hare, while no correlation was found with other species. In summary, this study found that salt block and feces induction could not improve the utilization rate of ungulates to underpasses of bundled linear infrastructure on Tibetan Plateau, and preliminary understood the factors affecting the utilization rate of underpasses.

Key words: Induction experiment, Qinghai-Tibet Plateau, railway ecology, road ecology, underpass, utilization rate, wildlife crossing structures

Introduction

Roads have become an important part of human society, with at least a quarter of the continental surface in Europe located within 500 meters of the nearest transport infrastructure (Torres et al. 2016; Medinas et al. 2019). However, while roads are beneficial to humans, studies have found that their impact on ecosystems is generally harmful (Krauss et al. 2010; Crooks et al. 2017; Barnick et al. 2022). For example, in Europe, an estimated 194 million birds and 29 million mammals die on the roads each year (Grilo et al. 2020). At the very least, Asia's roads threaten the survival and reproduction of Asian elephant (*Elephas maximus*), tiger (*Panthera tigris*), leopard(*Panthera pardus*) and Asiatic cheetah (*Acinonyx jubatus venaticus*) populations (UNEP/CMS 2019; Carter et al. 2020; Grilo et al. 2021; Dodd et al. 2024). Roads have a fragmenting effect on wildlife habitat and could reduce tiger populations worldwide by up to 20% (Carter et al. 2020); roads act as a barrier to communication among cougar populations, resulting in a decrease in genetic diversity (Riley et al. 2014); Wildlife crossing structures (WCSs) built to facilitate wildlife crossing roads also fail to achieve the desired effect of animal communication (Gloyne and Clevenger 2001; Rosell et al. 2023). A large number of studies have proved that roads will affect wildlife in terms of individual casualties, habitat loss, population isolation, etc. (Wang et al. 2013; Clements et al. 2014; Laurance et al. 2014; Fernandes et al. 2022; Sur et al. 2022). Understanding the impact of roads on wildlife is therefore important for biodiversity conservation (Forman and Alexander 1998; Li et al. 2019; Zhou et al. 2023).

The Tibetan Plateau region is known as the third pole of the Earth and an important biodiversity hotspot. The region is rich in wildlife resources, including rare species such as Tibetan antelope (*Pantholops hodgsonii*), wild yak (*Bos mutus*), kiang (*Equus kiang*) and snow leopard (*Panthera uncia*) (Li et al. 2018; Zhang et al. 2021). However, in recent years, with the increasing intensity of human activities and the continuous expansion of transportation infrastructure construction, wild animals are shrinking their range and are sometimes injured by breaking into human facilities (Kong et al. 2013; Dai et al. 2022; Lu and Huntsinger 2023).

In order to reduce the barrier effect of traffic facilities, WCSs have been widely used as a mitigation measure, aiming to provide a safe passage for wildlife to traverse transportation infrastructure and help maintain biodiversity and habitat connectivity (Sawaya et al. 2013; Seo et al. 2021; Helldin 2022). There are 33 specialized WCSs along the Qinghai-Tibet railway, with many multifunctional WCSs that wildlife may also utilize, which were put into operation on 1st July 2006 (Wu and Wang 2006). Studies of existing WCSs in the Tibetan Plateau showed that ungulates on the Tibetan Plateau initially avoided the crossing structures and had a low utilization rate (Bu et al. 2013); with the passage of time, they gradually adapted to and utilized the WCSs (Xia et al. 2005; Li et al. 2008; Wu et al. 2009; Zhang et al. 2009); different species have different adaptation cycles and learning curves to WCSs, and Tibetan antelope takes the longest adaptation time (Wang et al. 2021); Some passages are not used by wildlife because they are too close to human activity areas (Yin et al. 2006; Feng et al. 2013).

The cost of constructing a WCS is high and it is challenging to alter its location, size, or structure after installation. Therefore, it is important to establish methods for maximizing the effectiveness of WCS (Bond and Jones 2008; Downs and Horner 2012; Wang et al. 2019). Previous research conducted worldwide has focused on determining the factors that influence the efficiency of WCS, such as size (Forman 1998), traffic volume (Van der Ree et al. 2011), noise and light pollution (Denneboom et al. 2021), habitat corridor (Ceia-Hasse et al. 2017), and landscape characteristics (Ascensao et al. 2018). It is important to find ways to increase the utilization of WCSs and further reduce the impact of roads on wildlife in case of bundled infrastructure. However, there are few studies about it.

Previous research on underpasses along the Qinghai-Tibet railway and the Gonghe-Yushu expressway revealed a high utilization rate of small mammals, such as wolly hares and Tibetan foxes, while the utilization rate of ungulates was found to be relatively low (Wang et al. 2018). As a result, efforts are being made to develop strategies to enhance the utilization rate of ungulates. Most ungulates are social animals and some also have the habit of licking salt blocks (Razali et al. 2020; Maro and Dudley 2022). Therefore, we are placing salt bricks to provide the necessary food for ungulates, and also attempting to create a similar scent of their own kind through feces in animal corridors, in order to determine if these measures can improve the utilization rate of animal corridors.

Infrared camera technology, as a non-invasive, effective and reliable tool, is widely used in WCSs assessment (Burton et al. 2015; Barroso et al. 2023). It can record the behavior and activities of wildlife in the WCSs, capturing precious data that are difficult to obtain directly under human observation (Laidlaw et al. 2021; Schmidt et al. 2021). In this study, we have used infrared camera technology to evaluate utilization of underpasses on the Tibetan Plateau region, and tried to test the effects of salt bricks and feces on improving underpasses utilization. Through the results of this study, we hope to have an understanding of the utilization of the underpasses by wildlife in the Tibetan Plateau, and provide a scientific basis for the design and management of the underpasses in this region. This will help reduce the impact of human activities on wildlife, maintain ecological balance and biodiversity, and promote the sustainable development of the Tibetan Plateau region.

Methods

Study area

Two transportation corridors in Sanjiangyuan National Park are selected. The first corridor is the Qinghai-Tibet highway(G109) and railway transportation corridor, which passes through the Yangtze River Source Park of Sanjiangyuan National Park. The second is the Gonghe-Yushu expressway and highway(G214) corridor, which passes through the Yellow River Source Park of Sanjiangyuan National Park (Fig. 1a).

The Qinghai-Tibet railway and highway(G109) are bundled linear infrastructure. Built in the 1950s, the Qinghai-Tibet highway(G109) carries 85 percent of materials entering Tibet and 90 percent of materials leaving Tibet (Xia et al. 2007). The Qinghai-Tibet railway, which started construction in June 2001 and operated in July 2006, has become another major transportation artery connecting Qinghai province with Tibet Autonomous Region after the Qinghai-Tibet highway (Ge et al. 2011; Wang et al. 2017a). The Qinghai-Tibet highway(G109), with no fence, accommodates an average of 2,002 vehicles daily in 2023 at speeds not exceeding 80km/h. The Qinghai-Tibet railway operates an average of 30–40 trains per day in 2023, reaching a maximum speed of 100 km/h. It is fenced and features numerous underpasses. In the study area, highway and railway run parallel without intersecting, and there are no human activities such as grazing (Ru et al. 2018; Wang et al. 2018) (Fig. 1c).

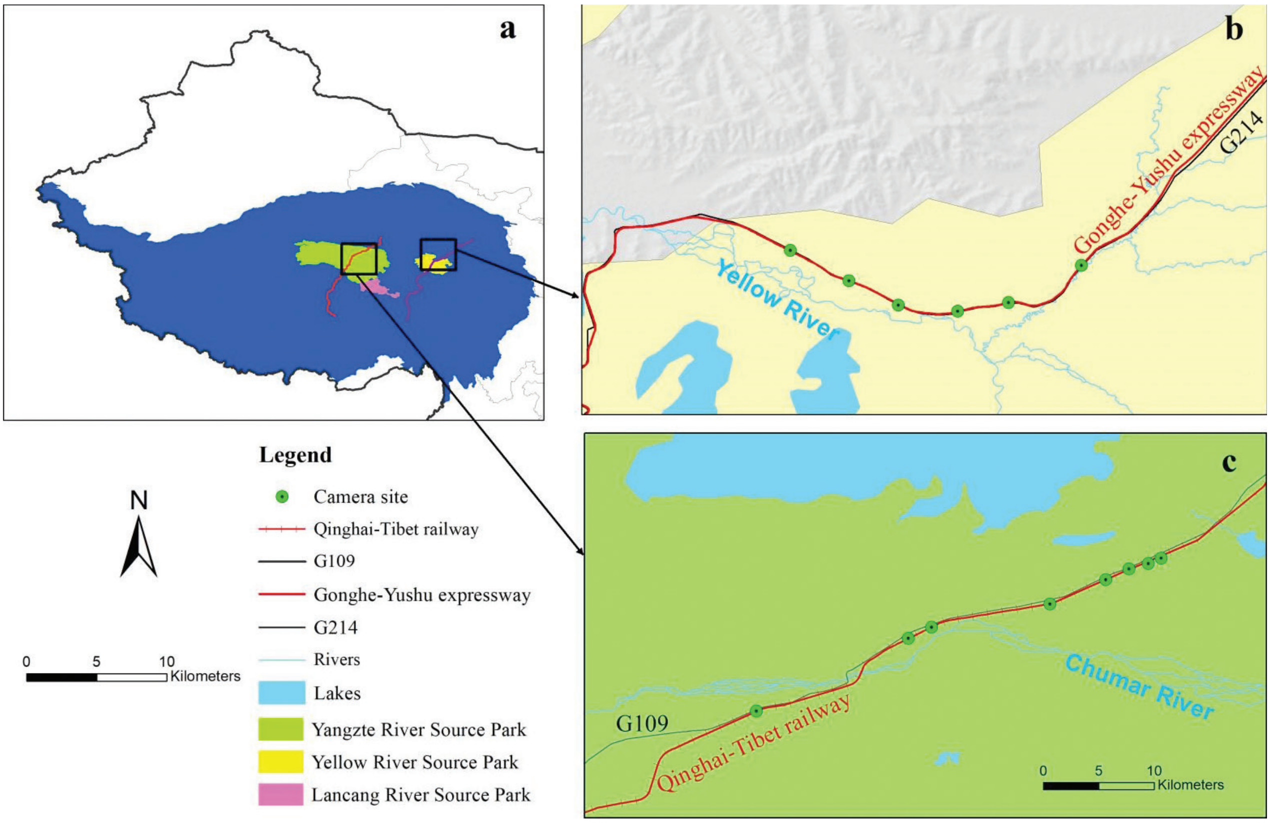

Figure 1. Schematic diagram of the study area and infrared camera sites **a** overall view of spatial relationship between two transportation corridors and Sanjiangyuan National Park (which includes Yangzte River Source Park, Yellow River Source Park and Lancang River Source Park) **b** Gonghe-Yushu expressway and highway research area and infrared camera sites **c** Qinghai-Tibet railway and highway research area and infrared camera sites.

The Gonghe-Yushu expressway and highway(G214) are also bundled linear infrastructure. The Gonghe-Yushu expressway operated in August 2017, becoming the first expressway in China to cross the permafrost region of the Tibetan Plateau. The Gongyu-Yushu expressway is entirely fenced and situated in grazing areas, leading to the construction of multiple underpasses to aid the movement of herders and animals. The maximum speed on this expressway is 100 km/h, with an average daily traffic of 1,800 vehicles. In contrast, Gonghe-Yushu highway(G214), which lacks fencing, sees an average of 1200 vehicles per day and has a specified speed limit of 80 km/h (Fig. 1b).

In the Qinghai-Tibet highway and railway transportation corridor, there are mainly 18 species of wild mammals living in the region. Including five species of national Class I protected, which are Tibetan antelope (*Pantholops hodgsonii*), wild yak (*Bos mutus*), kiang (*Equus kiang*), white-lipped deer (*Przewalskium albirostris*), snow leopard (*Panthera uncia*); Eight species of national Class II protected, which Tibetan gazelle (*Procapra picticaudata*), blue sheep (*Pseudois nayaur*), Tibetan argali (*Ovis hodgsoni*), Lynx (*Lynx lynx*), brown bear (*Ursus arctos*), grey wolf (*Canis lupus*) and Tibetan fox (*Vulpes ferrilata*) and red fox (*Vulpes vulpes*) (Yu et al. 2017; Xu et al. 2019). These animals have a wide range of distribution, and most of them have the characteristics of feeding, migration and breeding from low altitude to high altitude or from high altitude to low altitude with the change of season, and the migration needs to pass through the Qinghai-Tibet highway and railway transportation corridor. Among them, the long-distance seasonal migration characteristics of Tibetan antelope are the most typical, and they move upward in May-June every year and back migration in July-August (Lian et al. 2011).

In the Gonghe-Yushu expressway and highway corridor, the main animals along the expressway are the Himalayan marmot (*Marmota himalayana*), pika (*Ochotona curzoniae*), Tibetan gazelle, grey wolf, Tibetan fox, and kiang (Yang et al. 2020).

In summary, numerous wild animals inhabit both corridors, highlighting the conflict between transportation and wildlife.

Monitoring methods

We selected 8 and 6 small underpasses with similar dimensions and similar surroundings on the two transportation lines of Qinghai-Tibet railway and Gonghe-Yushu expressway, respectively, and set up an infrared camera for each small underpass (Ltl6310 wide angle; Shenzhen, China), adjusted the parameters and position to ensure that the field of view can observe the entire cross section in a complete and clear way, and left after turning on the camera. Along Qinghai-Tibet railway, over a 50-kilometer stretch, we identified 8 underpasses of similar size, each at least 1 kilometer apart (Fig. 2a). The dimensions of each underpass, including length, width, and height, are detailed in Table 1. Along Gonghe-Yushu expressway, over a 30-kilometer stretch, six underpasses of similar dimensions were chosen, each spaced at least 1 kilometer apart (Fig. 2b). The dimensions of these underpasses are detailed in Table 2. Notably, there are no intersections between the expressway and the highway within the study area, and human activities are limited to grazing (Wang et al. 2020). We set a salt block under 4 underpasses on Qinghai-Tibet railway and 3 underpasses on Gonghe-Yushu expressway each, and scattered the surrounding animal feces (Kiang, Tibetan antelope, and Tibetan gazelle feces were collected using a shovel while still fresh) as the experimental group. The other underpasses were left without any manipulation and served as the control group. We wrote warnings next to the infrared camera to avoid destruction or displacement, and explained the situation to surrounding residents. The distance between two adjacent infrared cameras was more than 1km, and the study period was from July 2022 to April 2023. The camera parameters were set as follows: The shooting mode was camera + video, the shooting interval was 1 minute, and 3 photos and 1 video (10 seconds) were shot in succession.

Data analysis

We identified mammals in the infrared camera photos, because the photos of animals other than mammals were not clear, so only mammals were analyzed statistically. Taking 30 minutes as an event, species that appeared repeatedly within a single event were only recorded as one time, which is a valid photo. At each camera site, we calculated the relative abundance index (RAI) for each species;

$$\mathrm{RAI} = \frac{\sum_{i=1} N_i}{\sum_{i=1} \mathrm{Trapday}_i}$$

Trapdayi is the number of days taken at camera site i, and Ni is the number of valid photos taken at camera site i of a particular species.

Table 1. Basic parameters of underpasses on Qinghai-Tibet railway.

Camera number	Experiment or control	Length/m	Width/m	Height/m	Openness Index	Distance from other road/m
1	control	8	16	5	10	1000
2	experiment	8	12	3.5	5.25	206
3	control	8	16	3.5	7	183
4	experiment	8	8	3.5	3.5	342
5	control	8	8	3.5	3.5	210
6	experiment	8	8	3.5	3.5	173
7	control	8	8	4	4	218
8	experiment	8	8	5	5	230

Note: Openness Index = Width × Height / Length.

Table 2. Basic parameters of underpasses on Gonghe-Yushu expressway.

Camera number	Experiment or control	Length/m	Width/m	Height/m	Openness Index	Distance from other road/m
1	control	30	4	3.5	0.47	44
2	experiment	30	4	3.5	0.47	40
3	control	30	4	3.5	0.47	50
4	experiment	30	4	3	0.40	41
5	control	30	4	3.5	0.47	38
6	experiment	30	4	3	0.40	48

Note: Openness Index = Width × Height / Length.

Figure 2. Photos of the underpasses **a** Qinghai-Tibet railway **b** Gonghe-Yushu expressway.

First, we counted the number of species appearing at each camera site, compared the number of species differences between the Qinghai-Tibet railway region and the Gonghe-Yushu expressway region, and the number of species differences between the experimental group and the control group in each study region. Secondly, we used Kruskal-Wallis test to analyze the difference of relative abundance index (RAI) of each species in the experimental group and the control group to judge the effect of salt block and feces induction experiment. Finally, using the " lme4" program package in R, we used the generalized linear mixed model (GLMM) by setting the length, width, height, and distance from the adjacent road of underpasses as fixed effect factors, and the two barriers (railway and expressway) as random effect to analyze the relative abundance index (RAI) of each species and the basic parameters in certain underpasses, and judge the relationship between the parameters of underpasses and the utilization intensity of species. All data analyses were carried out in R 4.1.2, with $p<0.05$ as the significant criterion.

Results

Overall species recorded in the underpasses

Among the 8 monitoring sites of the Qinghai-Tibet railway, we successfully recovered the infrared cameras of 7 monitoring sites, and the infrared camera No. 4 in the experimental group was lost for unknown reasons. In total 1,403 shooting events of wild mammals belonging to nine species were captured by the seven infrared cameras. These included wild yak, which are listed VU by the IUCN, and Tibetan antelopes, Tibetan gazelles and mountain weasels (*Mustela altaica*) listed as NT. Among the species with a high RAI were Tibetan antelope (RAI:0.3362), woolly hare (*Lepus oiostolus*) (RAI:0.2105), wolf (RAI:0.1604) and kiang (RAI:0.1076); Species with a low RAI are mountain weasels (RAI:0.0020) and lynx (RAI:0.0020) (See Suppl. materal 1: table S1).

We successfully recovered all infrared cameras at 6 monitoring sites set up in the Gonghe-Yushu expressway. The six infrared cameras captured a total of 319 shooting events of five wild mammals. Kiang, Tibetan fox, wolf, lynx and woolly hare photographed are all species listed as LC by the IUCN. Among them, the species with a high RAI are the Kiang (RAI:0.1084); and the species with a lower RAI is the lynx (RAI:0.0040) (See Suppl. materal 1: table S2).

Results of induction experiments

In the salt block and feces induction experiment in the Qinghai-Tibet railway area, it was found that the mountain weasels were only photographed in the underpasses of the control group, but not recorded in the underpasses of the experimental group. In addition, by comparing the RAI of each species in the infrared cameras of the experimental group and the control group, it was found that the eight species photographed by both the experimental group and the control group showed no difference between the two groups (Fig. 3; Suppl. materal 1: table S3).

The salt block and feces induction experiment in the Gonghe-Yushu expressway area found that the woolly hare was only photographed in the underpasses of the control group, but not recorded in the underpasses of the experimental group. In addition, by comparing the RAI of each species in the infrared cameras of the experimental group and the control group, it was found that the four species captured by both the experimental group and the control group showed no difference between the two groups (Fig. 4; Suppl. materal 1: table S4).

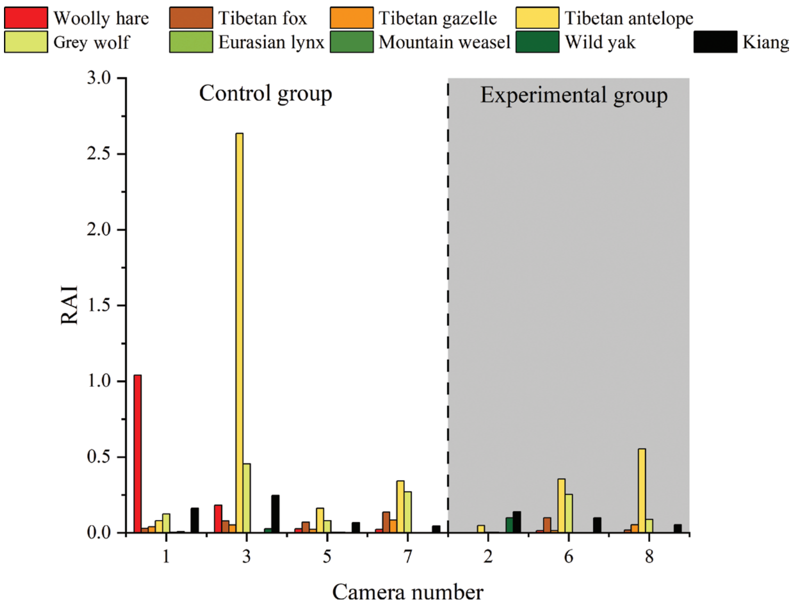

Figure 3. RAI of wild animals captured by infrared cameras on Qinghai-Tibet railway (Cameras 1, 3, 5 and 7 were the control group, 2, 6 and 8 were the experimental group, camera 4 was lost).

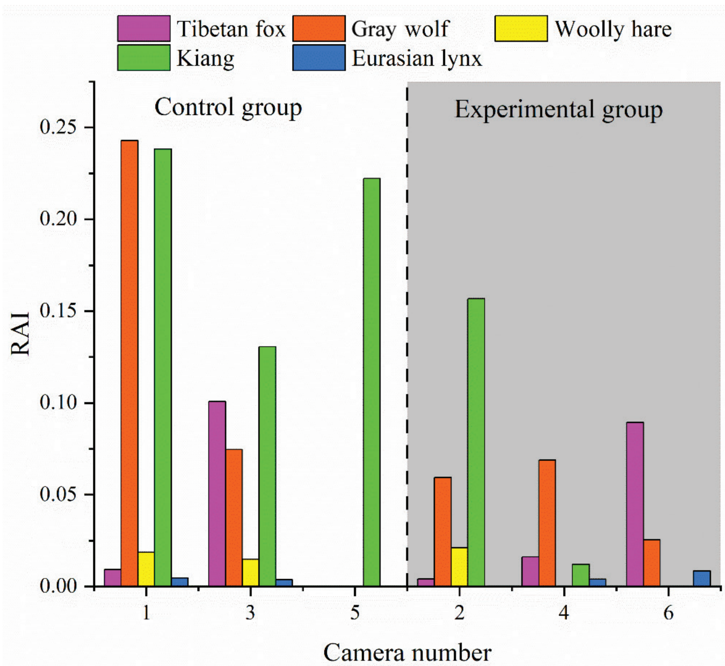

Figure 4. RAI of wild animals captured by Gonghe-Yushu expressway infrared camera (cameras 1, 3 and 5 were the control group, and 2, 4 and 6 were the experimental group).

Factors affecting underpasses' utilization by animals

By GLMM, we found that Tibetan antelope, Tibetan gazelle, Tibetan fox, Grey wolf, and Eurasian lynx did not show a correlation between the underpasses utilization and the basic parameters. The kiang showed that the longer ($z = 2.379$, $p = 0.017$) and wider ($z = 2.512$, $p = 0.011$) were the dimensions of the underpasses, the more frequently it appeared. Wolly hare showed a higher frequency of occurrence with longer ($z = 15.413$, $p < 0.001$) and wider underpasses ($z = 9.980$, $p < 0.001$), and greater distance from the adjacent road ($z = 14.848$, $p < 0.001$) (Table 3). Meanwhile, there was no difference in the relative abundance index (RAI) of the mountain weasel and wild yak in different underpasses, so no association with the basic parameters of the underpasses was analyzed (Table 3).

Table 3. GLMM between the relative abundance index (RAI) of each species and the basic parameters of underpasses in the infrared camera (p < 0.05 bold).

Species	Variables	*Z*	*p*
Tibetan antelope	Length	NA	NA
	Width	2.729	0.072
	Height	0.845	0.460
	Distance to other road	-2.168	0.119
Tibetan gazelle	Length	NA	NA
	Width	0.453	0.695
	Height	0.823	0.497
	Distance to other road	-0.687	0.563
Tibetan fox	Length	-1.200	0.230
	Width	-0.457	0.647
	Height	-1.192	0.233
	Distance to other road	0.210	0.833
Kiang	Length	2.379	**0.017**
	Width	2.512	**0.011**
	Height	0.596	0.551
	Distance to other road	-1.070	0.284
Woolly hare	Length	15.413	**<0.001**
	Width	9.980	**<0.001**
	Height	-1.247	0.212
	Distance to other road	14.848	**<0.001**
Grey wolf	Length	0.116	0.907
	Width	1.362	0.173
	Height	0.285	0.775
	Distance to other road	-1.092	0.274
Eurasian lynx	Length	0.298	0.765
	Width	0.004	0.996
	Height	-0.941	0.346
	Distance to other road	0.734	0.463

Discussion

Number of species differences in the study area of Qinghai-Tibet railway and Gonghe-Yushu expressway

Qinghai-Tibet railway falls under Yangtze River Source Park and the Gonghe-Yushu expressway falls under Yellow River Source Park of Sanjiangyuan National Park. Both of them belong to the alpine grassland ecosystem, the distribution of mammal species is very similar, and the species with higher and lower RAI values are similar, and both have relatively complete ecological chains. However, Tibetan antelopes, Tibetan gazelles, wild yaks and mountain weasels were found in the Qinghai-Tibet railway region, but not in the Gonghe-Yushu expressway region, indicating that the Yangtze River Source Park has a more complete ecosystem and better wildlife protection results than the Yellow River area. Among these four species, we have documented Tibetan gazelles and mountain weasels in the Gonghe-Yushu expressway area. The reason for not photographing them may be the high level of grazing activities along the expressway (Wang et al. 2020). However, there is almost no grazing activity in the current research area of the Qinghai-Tibet railway (Wang et al. 2018). Therefore, we urge for additional ecological protection to prevent the local extinction of these animals.

There is a significant amount of research indicating the impact of grazing on wildlife diversity (Waters et al. 2017; Pinto-Correia et al. 2018; Zhang et al. 2022). Similarly, studies in the Qinghai-Tibetan Plateau region have shown that increasing grazing intensity caused a decrease in biodiversity and ecosystem multifunctionality and that biodiversity and ecosystem function differed significantly between grazing intensities (Xiang et al. 2021; Liu et al. 2022; Liu et al. 2023). Therefore, it is crucial to focus on monitoring changes in grazing patterns in the study area. The construction of roads, which enhances transportation convenience and increases local grazing intensity, has been identified as a potential way in which roads can impact biodiversity.

Effect of salt brick and feces on inducing ungulates use in underpasses

We conducted salt brick and feces induction experiments on Qinghai-Tibet railway and Gonghe-Yushu expressway respectively, and the results showed that salt block induction experiments did not improve the utilization rate of underpasses in either of the two study areas. Our experimental results indicate that when the two underpasses are similar in size, salt brick and feces induction to attract ungulates that this does not improve the utilization rate of underpasses. The possible reason is that the soil on the Qinghai-Tibet plateau is salinized, and there are more ungulates licking the salt fields, and there is no shortage of salt (Zhang et al. 2012). The grasslands on both sides of the Qinghai-Tibet railway and the Gonghe-Yushu expressway have a lot of animal feces, so the feces at the entrance of the underpasses didn’t make any difference. In addition, the Qinghai-Tibet railway and Gonghe-Yushu expressway have been operated for 16 years and 6 years, respectively. Wildlife is likely to have adapted to the underpasses. We surmise that salt blocks and feces may be effective for newly built underpasses and may speed up the adaptation of wildlife to underpasses, but this needs to be tested in future new build underpasses.

Due to improper WCS positioning or inappropriate WCS size, many animal WCSs that have been built have not achieved the expected utilization effect

(Clevenger and Waltho 2005; Denneboom et al. 2021). However, it is difficult to modify WCSs after they are built, so it is meaningful to take measures to improve the utilization rate of WCSs. However, there is currently no well-developed technology for creating WCS habitats, and our experiment exploring the impact of salt bricks and feces on ungulates is not significant. Further research is needed to further reduce the impact of roads on biodiversity.

Effects of underpasses size on utilization

Previous research results show that the utilization rate of WCSs mainly depends on the size of the WCSs itself and the degree of human interference (Yin et al. 2006; Feng et al. 2013). The research findings indicate that Siberian roe deer (*Capreolus pygargus*) and wild boar (*Sus scrofa*) (Wang et al. 2017b), roe deer (*Capreolus capreolus*) and moose (*Alces alces*) (Bhardwaj et al. 2020), elk (*Cervus elaphus*) and deer (*Odocoileus* sp.) (Ng et al. 2004; Mata et al. 2008) all prefer wider WCSs. Similar results were found in this study, where we found that the wider the width of the underpasses, the higher the utilization rate of kiang and wolly hare. Therefore, when constructing new underpasses in our study areas, it is advisable to make them as wide as possible, provided that conditions allow.

In addition, this study also found that the farther the underpasses were from the adjacent highway, the higher the utilization rate of wolly hare. This result is consistent with previous studies showing that ungulates on the Qinghai-Tibet railway prefer short, wide and high underpasses and farther away from the road (Wang et al. 2018). These results suggest that when building underpasses, if there is a parallel road next to it, the more distance there is between the underpasses and the road, the better.

In this study, to ensure the comparative effectiveness of salt block and fecal induction, underpasses with similar basic parameters were selected. Therefore, the differences in variables such as length, width, height, and distance from the road are not large enough, which may be the reason why the number of species showing correlation is small. Additionally, the underpasses' utilization rate is also related to its location. The underpasses' utilization rate on animal dispersal routes is high, while the underpasses' utilization rate in areas with high human interference is low. These factors can result in narrow underpasses having a high utilization rate, and wide underpasses having a low utilization rate. The behavior patterns of different species can also lead to different preferences for animal pathways. Therefore, we should approach the conclusions of this paper with caution and carefully understand the local species situation when practicing in different regions to obtain a more effective method.

Conclusion

This study was the first to test the effect of salt brick and feces on improving the utilization rate of WCSs on the highways and railways of bundled linear infrastructure on the Tibetan plateau. We found that there are a large number of wild animals living along the Qinghai-Tibet railway and the Gonghe-Yushu expressway, and that the underpasses can be used. The kiang and wolf are the main species using the underpasses. The species of wild animals along the Qinghai-Tibet railway are more abundant than those along the Gonghe-Yushu expressway. We confirm that salt bricks and feces do not improve the utilization rate of underpasses significantly in Tibetan plateau. Finally, we observed that the incidence of wildlife use of the underpasses was related to the size and location of the passage itself, with wider underpasses and underpasses more isolated from other road disturbances being preferred by wildlife.

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Additional information

Conflict of interest

The authors have declared that no competing interests exist.

Ethical statement

No ethical statement was reported.

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Author contributions

Conceptualization, W.Y. and L.Z.; methodology, A.M. and W.Y.; software, A.M. and W.Y.; validation, A.M. and W.Y.; formal analysis, A.M. and W.Y.; investigation, A.M., W.Y and Y.Y.; resources, J.C., T.S., K.Y., and L.Z.; writing–original draft preparation, A.M.; writing–review and editing, J.C., Y.Y., T.S., K.Y., W.Y. and L.Z.; visualization, A.M.; supervision, W.Y.; project administration, L.Z.; funding acquisition, W.Y. and L.Z. All authors have read and agreed to the published version of the manuscript.

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Data availability

All of the data that support the findings of this study are available in the main text or Supplementary Information.

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Supplementary material 1

Supplementary information

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Data type: docx

Explanation note: **table S1.** Species of mammals that used small underpasses in Qinghai-Tibet railway area. **table S2.** Species of mammals that used small underpasses in Gonghe-Yushu expressway area. **table S3**. Kruskal-Wallis test results of RAI in the experimental group and control group in the Qinghai-Tibet railway region. **table S4.** Kruskal-Wallis test results of RAI in the experimental group and the control group in the Gonghe-Yushu expressway region.

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