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Screening for EIA in India: Enhancing effectiveness through ecological carrying capacity approach

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ABSTRACT

Developing countries across the world have embraced the policy of high economic growth as a means to reduce poverty. This economic growth largely based on industrial output is fast degrading the ecosystems, jeopardizing their long term sustainability. Environmental Impact Assessment (EIA) has long been recognized as a tool which can help in protecting the ecosystems and aid sustainable development. The Screening guidelines for EIA reflect the level of commitment the nation displays towards tightening its environmental protection system. The paper analyses the screening process for EIA in India and dissects the rationale behind the exclusions and thresholds set in the screening process. The screening process in India is compared with that of the European Union with the aim of understanding the extent of deviations from a screening approach in the context of better economic development. It is found that the Indian system excludes many activities from the purview of screening itself when compared to the EU. The constraints responsible for these exclusions are discussed and the shortcomings of the current command and control system of environmental management in India are also explained. It is suggested that an ecosystem carrying capacity based management system can provide significant inputs to enhance the effectiveness of EIA process from screening to monitoring.

1. Introduction

Around 110 low and middle income countries occupy about 75% of the world's land area and contain 93% of its population, but enjoy only about 19% of the world's 135 countries' gross domestic product (World Bank, 1997). Wood (2003) rightly pointed out that lack of improvement in EIA systems of developing nations will prove inadequate in terms of environmental protection at the global scale despite effective EIA systems in developed countries. The developing nations have also started some action in this regard as some 80 developing countries enacted some form of EIA legislation by the mid-1990s (World Bank, 1997; Glasson et al., 2005). However, the legal and institutional arrangements have not been made with a long term vision of sustainability. This shortcoming of EIA systems in most of the developing nations is being justified by citing their need to grow fast economically to be in a position to eliminate poverty and achieve the Millennium Development Goals (UN, 2005b).

The Screening guidelines for EIA reflect the first level of commitment the nation displays towards tightening its environmental protection system. This paper analyses the screening guidelines for EIA in India and dissects the rationale behind the exclusions and thresholds set in them. The screening process in India is compared with that of the EU EIA directive with the aim of understanding the extent of deviations from that of a screening philosophy in an economically developed scenario. Out of the exclusions from the EIA process in India the case of exemption for units inside industrial estates from the EIA process is analyzed in detail. The Tiruppur textile industry is highlighted for its positive and negative contributions and details about other similar cases are listed to show the extent of the problem. Finally the merits of ecological carrying capacity based environmental management are discussed along with its scope to improve the effectiveness of the EIA process from screening to monitoring.

2. Screening: significance in EIA

The process of screening can be defined as: "to determine whether or not a proposal should be subject to Environmental Impact Assessment (EIA), and if so, at what level of detail" (IAIA, 1999). Even though the above definition conveys the objective of

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the screening process in a simple and straightforward manner, the process of determining the same becomes complicated in a developing country. As this typically requires dedicated institutional capacity to carry out this task; time and resources from the project proponents; and imposes economic burden on small enterprises. EIA process at its best is expected to achieve the following: to protect the productivity and capacity of natural systems and the ecological processes which maintain their functions; and to promote development that is sustainable and optimizes resource use and management opportunities (IAIA, 1999). Although the stated objectives of every developing country is sustainable development (Earth Summit, 1997), in actual reality they are faced with complicated choices over conflicting tradeoffs primarily between the short term need to alleviate poverty and protection of environment for long term sustainability. Hence, the framing of screening guidelines assumes significance, as very stringent screening will hinder the economic growth of a nation and a liberal process or absence will result in inefficiency, wastage of resources and devastation of life-support systems (Jones, 1999; World Bank, 2002). Hence, the need for a rational screening process which even though need not stake out the exact mid point of the two probable scenarios outlined above, but has to tend towards a sustainable development strategy.

2.1. Approaches to screening

There are two fundamentally different approaches to EIA screening – (i) an environment centered approach, based on judgment of likely significant impact on environment from a proposal and (ii) a development-centered approach, based on the size and/or type of development. There is also a third option, being a hybrid of these two. Glasson et al. (2005) outlines the types of screening approaches as (i) thresholds approach and (ii) case-by-case screening approach. The thresholds approach adheres to the concept of a development approach where the thresholds are set in terms of size/capacity of projects and the case-by-case approach is an environment centered one where regardless of the size/capacity of proposals, every proposal is scrutinized for its impact on a specific environment. Lawrence (2003) details the screening approaches as being carried out in two stages such as determining (i) what should trigger an EIA requirement and (ii) which particular set of EIA requirements should be applied. Further, it is pointed out that these two screening steps normally focus on what (action), by whom (proponent) and where (the environment). Canter and Canty (1993) distinguish between screening based on policy delineation and that based on a preliminary study. Under screening based on policy delineation, further classification is made into project thresholds; sensitive area criteria; positive and negative lists.

An effective screening approach has to be a hybrid of the environment centered and development centered approaches. Because, though the environment in a certain location can be classified as sensitive, still certain benign green industries which can be located there needs to be listed for clarity and to avoid misuse or the other way the industries which should not be located can be specified. Such lists will remove the burden of case by case analysis on the screening authority and minimize the chance of ambiguity and corruption in decision making. Even in localities designated for location of all types of industries, only a certain quantity of pollutants can be safely assimilated by the various media (air, water and soil). Hence, a combined approach needs to be adopted to optimize the use of resources for effective screening. The details and the essential elements of such a combined approach are discussed in the forthcoming sections in the context of India.

2.1.1. Screening for EIA in India & the European Union

To evaluate a screening rationale compromised against the constraints of a developing nation, we need to analyze and compare it against a screening context free from most of those constraints. Hence, the screening guidelines of India are compared with that of the European Union with an aim to understand the extent of the deviations. The EU is characterized by high levels of literacy and per capita income than the developing world. High literacy levels ensure adequate environmental awareness and activism ensuring pressure on the authority to frame relevant legislation through adequate consultation with the stakeholders. High per capita income provides a country with resources to institute capacity aimed at environmental protection as indicated by the environmental kuznet's curve (UN, 2005a). The EU Directive is chosen for comparison as it can be considered to be free of the constraints faced by developing nations in general and India in particular. Moreover, the EU directive is a model legislation which needs to be emulated by the member nations and, hence, a better model to compare than the legislation of any country which invariably might be compromised by its prevailing socio-political regime.

In the EU, Environmental assessment is considered to be "a procedure that ensures that the environmental implications of decisions are taken into account before the decisions are made" (EU, 2005) and screening as "the process of determining whether or not EIA is required for a particular project" (European Commission, 2001). The screening requirement as per the EU Directives 85/337/EEC and 97/11/EC can be summarized as follow:

(i) Category of projects listed in Annex I of the EU directive which is well recognized as having the potential to affect the environment have to undergo EIA irrespective of their attributes.
(ii) The projects which under certain circumstances like low production capacity, location and technology might have negligible impact on the environment are listed in Annex-II of the directive. Whether the above premise holds well or not, has to be decided on a case to case basis guided by the criteria listed in Annex-III of this directive.

Though currently there is wide variation on thresholds and/or criteria adopted by EU member states, as per the directive, no project or activity can be excluded outright and at the least every project is required to be reviewed on a case by case basis against the specified criteria (European Commission, 2001). Now let us look at the developmental context and the screening rationale followed in India.

2.1.2. India: the screening context

India has invested considerable effort in carrying out the universally accepted principles of Rio Declaration. In one of its 27 principles, the Rio Declaration calls for EIA to be undertaken for activities that are likely to have a significant adverse impact on the environment (UN, 1992). The detailed analysis of India's EIA system under EIA Notification 1994 is available in Paliwal (2006) and Rajaram and Das, 2006. The screening guidelines of EIA 1994 are first presented and then the current guidelines as per EIA 2006 are discussed to understand the evolution of screening. Under EIA 1994, screening guidelines were issued for four categories of activities: Industry, Mining, Thermal Power, River Valley & Hydro-electric and Infrastructure. For the complete EIA notifications refer MEF (1994). The question of what will be put inside the EIA net and what will not be, evolved on an exclusionary non-participatory platform (Dubey, 2004).

The main approach to screening was one of excluding certain categories of projects based on investment thresholds. Hence, exclusions from EIA included industries in the small-scale sector (with an investment less than INR 10 million (Euro 0.2 million)), certain industrial projects with investment less than INR 1000 million (Euro 200 million). It can be presumed that SSIs were exempted for their role in poverty alleviation by employing unskilled labor in large numbers and other constraints which are discussed later in this paper. This provision of exclusions in EIA 1994 based on investments for both new and expansion projects had encouraged rampant 'salami slicing' by the project proponents to circumvent the EIA process in India (Kohli and Menon, 2005). Under this scenario, change towards the best practice screening was expected from EIA 2006 which is discussed next.

2.2. EIA Notification 2006: changes in screening requirement

The screening criteria for EIA Notification 2006 (EIA 2006 henceforth) were evolved by the MEF and though public comments were invited, only a select few of the interested groups were invited to express their opinion (Saldanha et al., 2007). The main change in the screening criteria of 2006 was the adoption of capacity based exclusions than the investment size of a project. Another key change is the division of projects into A and B categories based on capacity. The Ministry of Environment & Forests (MEF) deals only with category A projects and the State Environmental Impact Assessment Agency (SEIAA) under the State Pollution Control Boards (SPCB) screens the category B projects, classifies them into B1 and B2 (MEF, 2006). B1 projects require an EIA and B2 projects need only to submit information on Form-I (questionnaire requesting information on raw materials used, waste generated and environmental features of the location) along with an Environmental Management Plan (EMP) for emissions and effluents. Although these changes make the screening process similar to the Annex I & II projects of the EU directive, it has many deficiencies as detailed below.

2.2.1. Exclusions from screening in EIA Notification 2006: industries not listed in Schedule-I

The Annex I of EU EIA directive which list projects that have to carryout an EIA contains 22 types of projects with a total of 44 listings including the project sub-types. In comparison the Indian listing of category A projects in Schedule I of EIA 2006 contains 28 types of projects with a total of 34 including the sub-types. Further in comparison to 88 projects including sub-types in Annex-II of EU directive, there are only 34 projects including sub-types under category B in EIA 2006. This difference in the number of projects illustrates the extent of exclusions from the Indian EIA process. The projects listed in EU directive and excluded from Indian EIA are listed in Appendix 1.

The critical nature of this difference becomes clear when we consider the fact that owing to high population and an environment dependent majority, impacts considered negligible in EU will have a substantial effect on the environment and communities in India (refer Rajaram and Das, 2007 for detailed discussion). This would mean that the listing of activities for screening in developing nations such as India has to be much more comprehensive than that of the EU. Moreover, the criteria for screening category B projects for further classification into B1 and B2 have not been specified till date.

2.2.2. Capacity based exclusions of listed industries/projects

For many of the projects which are covered in EIA 2006, capacity thresholds have been specified below which they are excluded from the EIA requirement. These projects along with the exclusionary threshold/criteria are listed in Table 1. From the above listing it can be seen that all the excluded capacities have the potential to impact the environment if located in ecologically fragile areas and ecosystem dependent communities. The philosophy behind these exclusions might be the aim to lessen the burden on the proponents and authorities rather than effective environmental protection. It can be noticed that industrial estates have been given a major concession. It is obvious that this is aimed at encouraging industrial growth, but the track record of industrial estates in adhering to the environmental norms are very poor (Polluted Places, 2008; Greenpeace, 1999; Banerjee, 2003; Rajaram and Das, 2008a). Since the EIA 2006 does not list all industries with impact potential like the EU annex II, it is possible to setup an industrial estate of less than 500 Ha area (say 490 ha) comprising entirely of small-scale textile dyeing units without carrying out an EIA. Such industrial estates which are functioning presently have had significant negative impact in the environment as detailed in the next section.

Table 1
List of projects with exclusionary thresholds in EIA 2006

Ref no.	Type of Project	Capacity/Criteria
1a	Mining	<5 Ha
1c	Hydroelectric power plants	<25 MW
1d	Thermal power plants Pet coke diesel and other fuels	<5 MW
3a	Non-toxic secondary metallurgical processing	<5000 ton/annum
4b	Coke oven plants	<25000 ton/annum
4d	Chlor-alkali (membrane tech)	inside industrial estates
4f	Leather/hide/skin processing	inside industrial estates
5e	Petrochemical based processing	inside industrial estates
5f	Synthetic organic chemicals	inside industrial estates
5j	Sugar industry	<5000 ton/day cane crushing capacity
7c	Industrial estates/EPZ/SEZ/Biotech parks/leather complexes	<500 Ha and not having any category A or B industry
7e	Ports & harbours	<10000 ton/annum of fish handling
7f	Highways	expansion for < 30 km
8a	Building & construction projects	<20000 sq.m.
8b	Townships and area development projects	<50 Ha & <150000 sqm

3. Blanket exemption for certain industries in industrial estates – why and where it can lead?

Under EIA 1994, the Small Scale Industry (SSI) was an outright exemption from EIA and in EIA 2006 they are still given the concession if they are located inside industrial estates. The case of exclusion of SSI is taken up for further discussion and analysis mainly because of the impact they have had on the environment which can be reduced through their inclusion into the EIA system. The reasons put forward in India for concession to SSIs are as follow: small investors cannot spend for EIA studies; the quantity of pollutant release is small when compared to large factories; EIA and EC delays the process of setting up an industry; the negative impact on the environment is small when compared to the bigger positive impact of job creation; the impact of pollution is local and can be monitored by State Pollution Control Boards (SPCB); excluded SSIs can be put in industrial estates and facilitated through common effluent treatment plants (CETP) and industrial growth is the only way to achieve poverty alleviation.

The role of SSIs – which are similar in nature to certain extent to the TVIEs (Town & Village Industrial Enterprises) in China, Small and Medium Enterprises (SMEs) as they are termed in Europe and Small Businesses in US – in the economic growth in general and job creation in particular is well appreciated and

Governments have initiated special laws to consider their interests in environmental law and enforcement (Agarwal, 2001; ECOTEC, 2000; EPA, 2005). The impact of SSIs on the economy can be clearly understood when we consider that it provides immediate large scale employment to unskilled workforce, offering a method of ensuring a more equitable distribution of national income and facilitating an effective mobilization of resources of capital and skill which might otherwise remain unutilized (Gulati, 1996: pp1). In India the SSIs together with Micro and medium enterprises have a share of 40% in the industrial production and 33% of the total manufactured exports employing about 31 million people in 12.8 million enterprises. The labor intensity in this sector is estimated to be four times higher than larger enterprises (MoMSME, 2006). The World Bank and the International Finance Corporation (IFC) have been particularly active in promoting small-scale enterprises, setting up a separate department for them in 2000 and allotted USD1.5 billion toward their development in 2002 (Rajshri and Lanjouw, 2004). The environmental degradation associated with uncontrolled promotion of SSIs is also widely recognized (ECOTEC, 2000; Snigdha and Mitrab, 2005; Ogenis, 2001). The case of Tiruppur textile industry is presented in the next section to illustrate their positive economic impact and the ineffectiveness of the current strategies in controlling their negative impact on the environment.

3.1. Case study: The Tiruppur textile industry

3.1.1. The Tiruppur textile industry: economic contribution

Tiruppur, the leading cotton knitwear industrial cluster in South India, located in Tamil Nadu State has more than 9000 small-scale knitwear related units employing about 500,000 people. The export valued from Tiruppur during the year 2006–7 was about USD 2 billion (Samuel Raja, 2008).

3.1.2. Impact on Environment

The study by Appasamy and Nelliyat (2000) brought out the following facts about the environmental impacts in Tiruppur: 702 bleaching and dyeing units were functioning by 2000 and their water consumption was about 86 million litres per day (MLD). Despite the construction of individual and common effluent treatment plants at considerable cost, salts, mainly chloride, continue to be discharged unabated. Although each individual unit discharges only a small quantity of effluents, the combined discharge of more than 700 bleaching and dyeing units outstrip the assimilative capacity, causing damage to agriculture, fisheries, and local ground water in and around Tiruppur.

3.1.3. Judicial Intervention

The farmers got themselves organized and resorted to agitation and legal recourse demanding the judiciary to rectify the situation brought on by the failure of the Government. The judiciary promptly pulled up the SSIs for not heeding their earlier directions and ordered them to pay up the subscription fees for putting up a joint zero discharge effluent treatment plant within a deadline or face closure (Sridhar, 2005a). The 'Loss of Ecology Commission' a State Government Agency had asked the Tiruppur dyes union to pay INR 4 Crores (USD 0.83 million) of compensation, a figure contested by the farmers union as it works out to a meagre INR 240/hectare (Sridhar, 2005b). As of June 2009, the Tiruppur SSIs were lobbying the government to implement a 300 km effluent pipeline project estimated to cost INR 800 Crores to convey the effluent to the Bay of Bengal (BS, 2009).

3.2. Is Tiruppur an isolated case?: How industrial estates are Responding to central control

The environmental damage perpetuated by the textile industry in Tiruppur is not an isolated case as evident from the environmental damage in these industrial estates dominated by SSI clusters: Ankleshwar-Chemicals, Howrah-Foundries, Kanpur-Tanneries, Nandesari-Chemicals, Panipat-Chemicals, Ambur-Tanneries, etc. (Polluted Places, 2008; Greenpeace, 1999; Banerjee, 2003; Rajaram and Das, 2008a). For more details about other sites in India devastated by SSIs see Polluted Places (2008). In the light of the contributions and problems associated with the SSIs, how many and how much of the arguments put forward for their exclusion is valid? Is there any way to move forward and progress towards sustainable industrial growth and poverty alleviation without its attendant destruction of life-support systems? Can EIA play any role at all in this constraint ridden situation? To understand the reasons for this situation it is necessary to look at the overall framework of environmental management in India and the position occupied by screening for EIA in it.

4. The link between screening for EIA and general pollution control in India

The link between the environmental management of projects which are required to conduct an EIA and those that are exempted from an EIA is given in Fig. 1. The figure shows that, category A industries which are required to conduct an EIA as per EIA 2006 are dealt by the MEF for grant of environmental clearance and the SPCB is required to conduct the public hearing and forward the minutes along with the final EIS to MEF. Whereas category B industries go through the screening process by the SPCB and if these projects are categorized as B1 (EIA required) go through all the steps as a category A industry but at the state level with the SPCB being the sole clearance authority. The projects categorized as B2 (EIA not required) are required to submit an environmental management plan and apply for 'consent to establish (CTE)' under the Air and Water Acts. The SPCB scrutinizes the EMP and provides CTE with or without conditions. The construction of the project is started and when it is ready for commissioning, they need to apply to the SPCB for 'consent to operate (CTO)'. The SPCB verifies the implementation of the EMP and provides CTO with which the project can be commissioned.

4.1. Why EMP is inadequate for Unlisted industries

Industries which are exempted from conducting EIA studies are required to apply for CTE by filling up forms under The Water (Prevention and Control of Pollution) Act, 1974 and The Air (Prevention and Control of Pollution) Act, 1981 (MEF, 2010). These forms typically request information regarding the raw materials used, quantity of water consumed, wastes generated (air emissions, liquid effluents, solid/hazardous wastes) and the treatment scheme proposed for treatment and disposal of the wastes. The EMP typically consists of proposed treatment schemes for disposal of the contaminants in concentrations upto or below the allowable limit specified by the SPCBs. These limits of contaminants in India are collectively known as Minimum National Standards (MINAS). For example as per MINAS, liquid effluents with Bio-chemical Oxygen Demand of less than 30 mg/L ($BOD<30$ mg/L) can be discharged into water bodies, provided the other contaminants listed under MINAS also meet their limits. These uniform discharge standards such as MINAS are ineffective in controlling pollution of the environment as they do not consider the cumulative effect of high

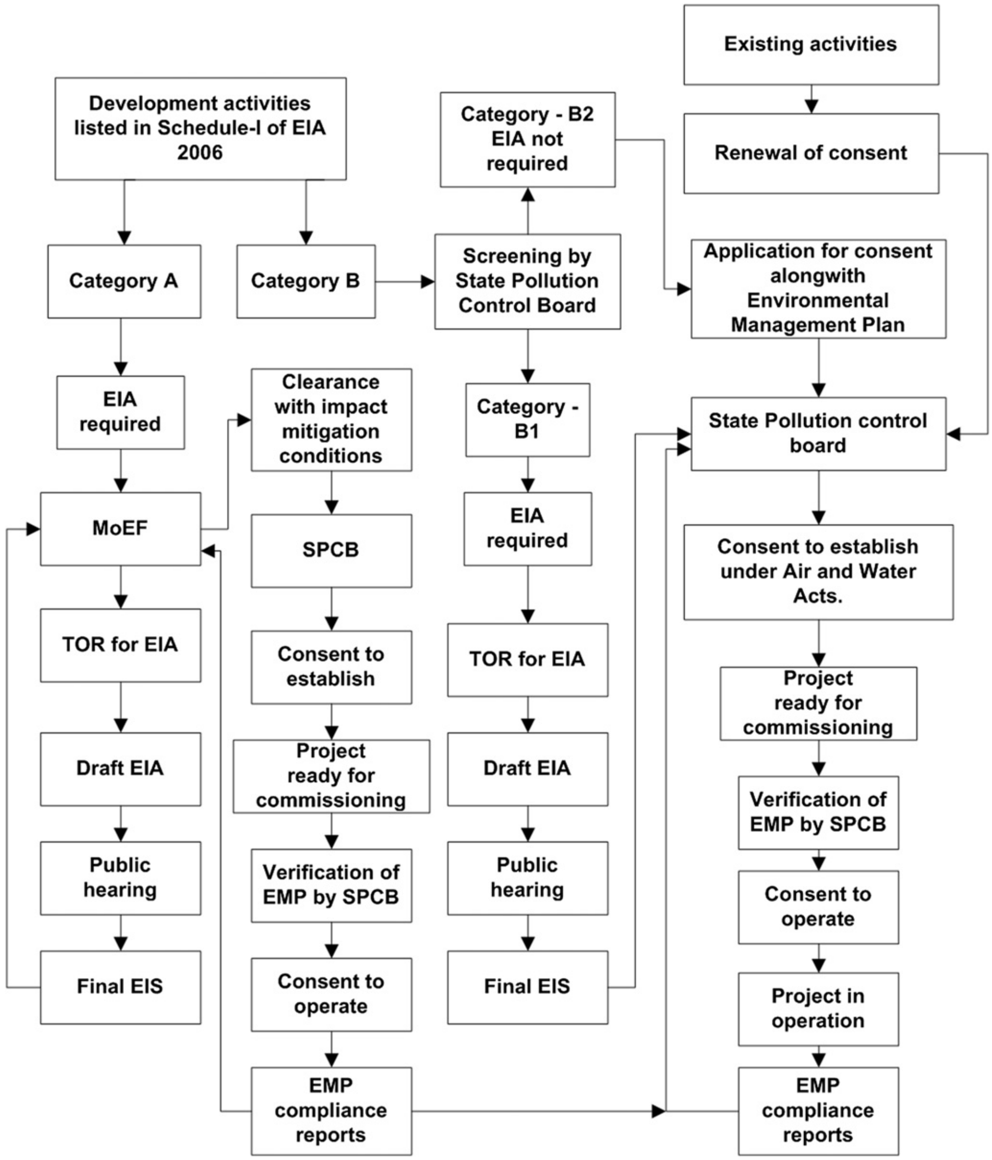

Fig. 1. Flow sheet of environmental management for projects requiring EIA and projects exempted from EIA.

volume of discharges from single or multiple sources. Hence, developed nations have already adopted ecosystem specific standards such as Total Maximum Daily Load (TMDL) (USEPA, 2008). The detailed discussion of the drawbacks of the MINAS under the Command and Control (CAC) system for control of Industrial effluents in India can be found in Rajaram and Das (2008a).

4.2. How cumulative impacts are ignored under mere environmental management plans (EMPs)

Exemption of industries from the EIA process under the premise that they can be taken care through the consent forms has not proved to be the right policy. Even industries conducting EIA studies still submit an EMP for liquid effluents and air emissions aimed at satisfying the MINAS and not the ecosystem specific impact mitigation plan. This compromises the effectiveness of the whole EIA system and reduces it to a mere form filling formality in India. When the consideration of direct impacts from activities is not mitigated through the EMP and its effective implementation and follow-up, consideration of cumulative impacts under the EMP remains elusive in India. But as calculating the cumulative impacts require data from multiple activities, it is the regulatory authority which is best placed to carryout the task than the proponent of a single activity.

5. Discussions: carrying capacity based clearances as an Alternative to Conventional project EIAs

The problem of how to bring all the industries including the SSIs into the EIA net can be solved if the constraints regarding access to expert knowledge and cost of conducting the study can be reduced and/or shared. Traditionally as per the EIA systems which evolved in the developed countries, the project proponents are responsible to carryout EIA at their expense. This cost which works out to be a fraction of the total investment for a large scale venture assumes a larger proportion for smaller ventures i.e. SSIs/SMEs. Hence, instead of excluding the SSIs from conducting such studies, why should not the regulatory authority take the responsibility? The chronic problem of credibility of EIA studies conducted by the consultant-proponent nexus can be cured by shifting the responsibility of determining the significance of impacts to the regulatory authority with involvement of local public/NGOs. The model where the Government takes responsibility to carry out EIA and achieve its intended objectives (to protect the productivity and capacity of natural systems) by linking it with ecological carrying capacity of the area is proposed in Fig. 2.

This proposed model requires the following to be effective: detailed database of ecological processes and the functions they maintain, carrying capacity of the natural system in terms of its productivity and safe pollutant assimilation, linkages which the local populace has with ecological components and current status/trend of key resources in terms of its sustainability. The regulatory authority with the help of such information will be in a good position to judge the impact of any new activity on the sustainability of the local human–ecological interactions. Moreover, industries affect the environment mainly through extraction of resources (water chiefly), discharge of liquid effluents, and emission of air pollutants and disposal of solid/hazardous wastes. Other impacts in the case of SSIs are minor in comparison to large projects and are negligible when located in urban/industrial areas. Of the four main impacts outlined above, except solid/hazardous waste other impacts cannot be transferred to other ecosystems easily. And in terms of their effect on the local ecosystems, air emissions have the least impact when compared to other factors. This is a key factor in the unrestrained release of greenhouse gases which have impacted the global commons (atmosphere). Hence, extraction of resources and discharge of water pollutants which have a cumulative impact on the local ecosystem have to be regulated based on the carrying capacity. The solid/hazardous waste needs to be integrated into the regional waste management plan and the air emissions have to be tied along with the national emissions target.

For any EIA system to be successful it should have an efficient monitoring programme to ensure that pre-project plans & promises are met consistently. It is important that the EMP from EIA is integrated properly into the EMS of the functioning project for it to be effective (Bailey, 1997; Morrison-Saunders and Bailey, 1999). In resource scarce developing nations, this cannot be achieved by SPCBs alone without the participation of local community. Industrial pollution reduction through informal regulation by community pressure has been recognized as being influential (Schumacher, 1989; Blackman and Bannister, 1996; Goldar and Banerjee, 2004). The local community in many parts of the world which is primarily dependent on the ecosystems through agriculture and/or hunting-gathering have a wealth of knowledge about their environment. This ability to read local ecological processes mainly developed through experience and passed on through generations by their instincts to just survive. The model of community based resource management which is popular in the area of forest management (Nayak and Berkes, 2008) needs to be replicated in other ecosystems as well. Hence, we have to institutionalise their role as partners in managing our ecosystems with proper consideration and utilization of traditional ecological knowledge for fixing criteria and indicators of ecosystem health apart from the use of resource intensive scientific models (Rajaram and Das, 2008b).

Further, the Government needs to strengthen its SPCBs in terms of manpower and infrastructure as more investment is likely in future in the dirty sectors of chemicals, pesticides, and every imaginable industry with high pollution potential. The cost of carrying out all these studies need not be borne fully out of public funds. Instead, the total cost can be divided among the SMEs which will definitely be lesser than the cumulative cost when the SMEs conduct them individually and can also be collected in monthly or quarterly instalments much like the yearly fees for license under the Water and Air Acts collected from them at present. Funds spent on such ventures are justified when we consider the fact that annually ecosystem degradation in India is estimated to be around 10% of GDP or around USD 70 billion (Pachauri, 2004) (based on GDP of USD 688.7 billion in 2004 (World Bank, 2005)) which is the loss of capital asset and much of it irreversible for a long time.

From the proposed Environmental clearance procedure the regulating authorities with the help of local participation can keep account of every effluent outlet and control the industrial development of any area inline with its carrying capacity as staying within source and sink capacity is primary to achieve sustainable development (Sadler, 1996, pp 209). Of course SMEs approved through this system can still go on to release untreated effluents, but monitoring and control through local public participation can avoid the repeat of Tiruppur like situations in future and promote pro-poor growth of SMEs which ultimately will lead to alleviation of poverty, the important of Millennium Development Goals. The cost of developing local ecosystem databases, conducting EIA and carrying capacity studies can be further reduced by involving NGOs, research institutes and local universities and can be made more significant and participatory by incorporating Local Ecological

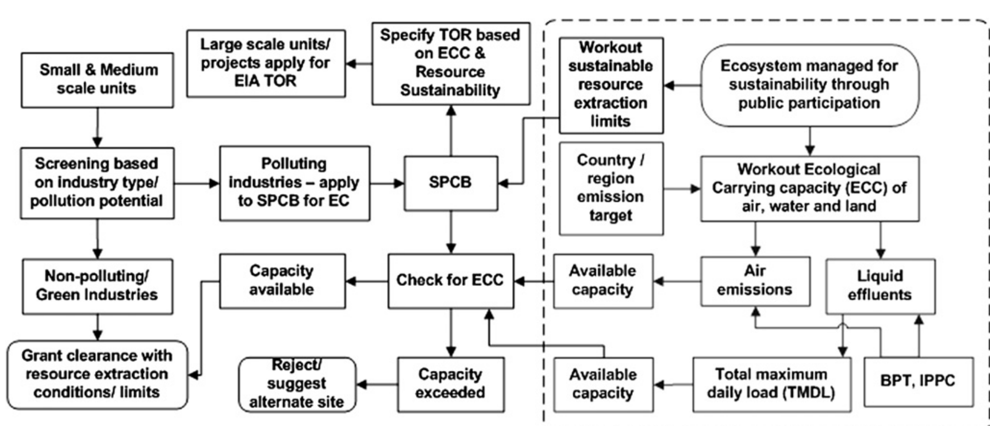

Fig. 2. Modified Environmental Clearance & Monitoring Process based on Ecological Carrying Capacity.

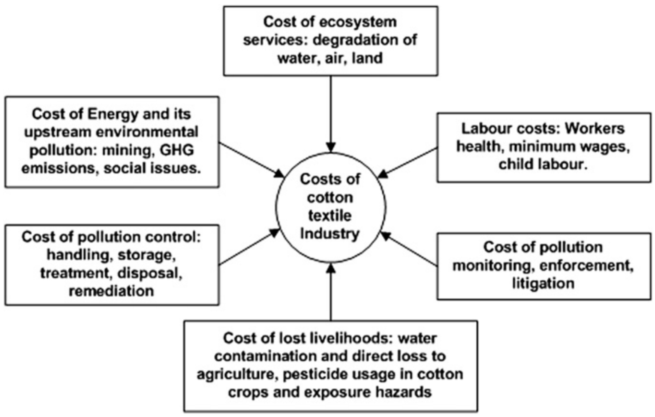

Fig. 3. Costs of Textile Industry which are to be internalised.

Knowledge (LEK). This approach will definitely lead to a much effective system of environmental conservation and management as an informed citizenry is recognized of having the ability to reduce environmental disasters (Skanavis et al., 2005).

5.1. *Economic growth Led poverty alleviation at what cost?*

The task of designing environmental management systems cannot be undertaken blindly to fulfil the requirements of economic growth without scrutinizing the actual need/attendant effects of unrestrained economic growth. Globally it has become very clear that liberalised economic growth aimed at creating wealth which is expected to trickle down to the poor is not happening but is increasing poverty and inequality (Stiglitz, 2002). Further it is the poor who suffer most due to ecosystem degradation (Millennium report, 2005) which in the case of Tiruppur SSIs has been the result of externalizing the cost of economic production & export of textiles. To arrive at the true cost of production of textiles in this case, we need to account for the various components of cost of ecosystem damage into the cost of textiles manufactured which is presently borne collectively by the society as shown in Fig. 3. According to a World Bank study, between 1975 and 1995, as India's GDP doubled, industrial and vehicular pollution load went up between four and eight times respectively (Anon, 1999, pp 32). Deterioration in urban environment, increase in slum population, and in air, river, and water pollution has vastly affected the quality of life of the urban poor (Khurana, 2004, pp1).

The trend of relying on exports like textiles, pesticides, chemicals and other products which the developed world is willing to import (for the simple reason that the real cost of manufacturing them are unrecoverable and unjustifiable against the irreversible loss of life-support systems) is not sustainable in the long term. Hence, there is an urgent need for any society to adopt a zero tolerance policy when it comes to safeguarding its life-support systems for continued sustainable survival. But even pollution intensive Industrial manufacturing is very much required for satisfying any society's internal consumption, and to trade the surplus produced according to sustainable strategies in order to import goods and services which are locally unavailable.

Implementing sustainable strategies should begin through existing tools such as EIA and focus on making it effective. Sadler (1996, pp39) identifies three distinct review parameters as: procedural: – does the EA process conform to established provisions and principles?, substantive: – does the EA process achieve the objectives set, i.e., support well informed decision making and result in environmental protection? and transactive: – does the EA process deliver these outcome at least cost in the minimum time possible, i.e., is it effective and efficient?. The substantive objective of environmental protection can only come about through an EIA system with components from screening to monitoring strengthened to enable development which is environmentally sustainable. A supporting system such as an ecosystem carrying capacity based management system can provide significant inputs to enhance the effectiveness of EIA process from screening to monitoring.

6. Conclusions and Recommendations

The current screening regulation of EIA 2006 in India with its exclusions is off-line from a sustainable development strategy. A more logical inclusive approach along the lines of the EU directive should be adopted. The list of projects in category B needs to be expanded to include a number of projects clearly identified in Annex II of the EU EIA directive. Clear and transparent criteria for categorizing projects into B1 and B2, needs to be specified in EIA 2006. The SSIs/SMEs in industrial estates excluded from EIA system have polluted the ecosystems around industrial areas across the country threatening India's sustainability and need to be brought under the EIA system. The constraints of the SSIs can be alleviated by adopting the proposed EIA system based on ecological carrying capacity where their constraints are considered. The Project EIA system for industries needs to be integrated with SEA and carrying capacity studies with SPCBs and local institutions playing the central responsible role in pre-project EIA and post-clearance monitoring. The SPCBs in India need to be strengthened in terms of infrastructure and manpower to protect the environment against the increasing tide of polluting industries which will be setup in India in the coming years. Framework and methods to incorporate the local ecological knowledge of the local people must be developed and adopted in the EIA process to enable it to achieve its substantive purposes.

Appendix. Projects with impact potential not covered under EIA 2006 in comparison with EU directive.

Annex I (EIA Mandatory)

7. (a) Construction of lines for long-distance railway traffic 11. Groundwater abstraction or artificial groundwater recharge schemes where the annual volume of water abstracted or recharged is equivalent to or exceeds 10 million cubic metres. 13. Waste-water treatment plants with a capacity exceeding 150 000 population equivalent as defined in Article 2 point (6) of Directive 91/271/EEC. 17. Installations for the intensive rearing of poultry or pigs with more than: (a) 85 000 places for broilers, 60 000 places for hens; (b) 3000 places for production pigs (over 30 kg); or (c) 900 places for sows. 20. Construction of overhead electrical power lines with a voltage of 220 kV or more and a length of more than 15 km.

Appendix (*continued*)

Annex II (to be screened for EIA)

1. Agriculture, Silviculture and aquaculture (a) Projects for the restructuring of rural land holdings; (b) Projects for the use of uncultivated land or semi-natural areas for intensive agricultural purposes; (c) Water management projects for agriculture, including irrigation and land drainage projects; (d) Initial afforestation and deforestation for the purposes of conversion to another type of land use; (e) Intensive livestock installations (projects not included in Annex I); (f) Intensive fish farming; (g) Reclamation of land from the sea. *2. Extractive Industry* (a) Quarries, open-cast mining and peat extraction (projects not included in Annex I); (b) Underground mining; (c) Extraction of minerals by marine or fluvial dredging; (d) Deepdrillings, in particular: – geothermal drilling, -drilling for water supplies, with the exception of drillings for investigating the stability of the soil; *3. Energy Industry* (i) Installations for the harnessing of wind power for energy production (wind farms). *4. Production and processing of metals* (a) Installations for the production of pig iron or steel (primary or secondary fusion) including continuous casting; (b) Installations for the processing of ferrous metals: (i) hot-rolling mills; (ii) smitheries with hammers; (iii) application of protective fused metal coats; (c) Ferrous metal foundries; (d) Installations for the smelting, including the alloyage, of non-ferrous metals, excluding precious metals, including recovered products (refining, foundry casting, etc.); (e) Installations for surface treatment of metals and plastic materials using an electrolytic or chemical process; (f) Manufacture and assembly of motor vehicles and manufacture of motor-vehicle engines; (g) Shipyards; (h) Installations for the construction and repair of aircraft; (i) Manufacture of railway equipment; (j) Swaging by explosives; (k) Installations for the roasting and sintering of metallic ores. *5. Mineral Industry*

(d) Installations for the manufacture of glass including glass fibre; (e) Installations for smelting mineral substances including the production of mineral fibres; (f) Manufacture of ceramic products by burning, in particular roofing tiles, bricks, refractory bricks, tiles, stoneware or porcelain. *6. Chemical Industry (not included in Annex I)* (a) Treatment of intermediate products and production of chemicals; (b) Production of pesticides and pharmaceutical products, paint and varnishes, elastomers and peroxides; *7. Food Industry*

(a) Manufacture of vegetable and animal oils and fats; (b) Packing and canning of animal and vegetable products; (c) Manufacture of dairy products; (d) Brewing and malting; (e) Confectionery and syrup manufacture; (f) Installations for the slaughter of animals; (g) Industrial starch manufacturing installations; (h) Fish-meal and fish-oil factories; (i) Sugar factories. *8. Textile, Leather, wood and paper products* (a) Industrial plants for the production of paper and board (projects not included in Annex I); (b) Plants for the pretreatment (operations such as washing, bleaching, mercerization) or dyeing of fibres or textiles; (d) Cellulose-processing and production installations. *9. Rubber Industry* - Manufacture and treatment of elastomer-based products. *10. Infrastructure projects* (c) Construction of railways and intermodal transshipment facilities, and of intermodal terminals (projects not included in Annex I); (e) Construction of roads, harbours and port installations, including fishing harbours (projects not included in Annex I); (h) Tramways, elevated and underground railways, suspended lines or similar lines of a particular type, used exclusively or mainly for passenger transport; (k) Coastal work to combat erosion and maritime works capable of altering the coast through the construction, for example, of dykes, moles, jetties and other sea defence works, excluding the maintenance and reconstruction of such works; (l) Groundwater abstraction and artificial groundwater recharge schemes not included in Annex I; *11. Other projects* (a) Permanent racing and test tracks for motorized vehicles; (b) Installations for the disposal of waste (projects not included in Annex I); (c) Waste-water treatment plants (projects not included in Annex I); (d) Sludge-deposition sites; (e) Storage of scrap iron, including scrap vehicles; (f) Test benches for engines, turbines or reactors; (g) Installations for the manufacture of artificial mineral fibres; (h) Installations for the recovery or destruction of explosive substances; (i) Knackers' yards. *12. Tourism and leisure* (a) Ski-runs, ski-lifts and cable-cars and associated developments; (b) Marinas; (c) Holiday villages and hotel complexes outside urban areas and associated developments; (d) Permanent camp sites and caravan sites; (e) Theme parks. *13 – Any change or extension* of projects listed in Annex I or Annex II, already authorized, executed or in the process of being executed, which may have significant adverse effects on the environment; – Projects in Annex I, undertaken exclusively or mainly for the development and testing of new methods or products and not used for more than two years.

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Conservation Biology

REVIEW

Animal agency in wildlife conservation and management

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Article impact statement: Incorporating animal agency into wildlife conservation and management can lead to more effective, nuanced, and just outcomes.

Abstract

Wildlife conservation and management (WCM) practices have been historically drawn from a wide variety of academic fields, yet practitioners have been slow to engage with emerging conversations about animals as complex beings, whose individuality and sociality influence their relationships with humans. We propose an explicit acknowledgement of wild, nonhuman animals as active participants in WCM. We examined 190 studies of WCM interventions and outcomes to highlight 3 common assumptions that underpin many present approaches to WCM: animal behaviors are rigid and homogeneous; wildlife exhibit idealized wild behavior and prefer pristine habitats; and human–wildlife relationships are of marginal or secondary importance relative to nonhuman interactions. We found that these management interventions insufficiently considered animal learning, decision-making, individuality, sociality, and relationships with humans and led to unanticipated detrimental outcomes. To address these shortcomings, we synthesized theoretical advances in animal behavioral sciences, animal geographies, and animal legal theory that may help conservation professionals reconceptualize animals and their relationships with humans. Based on advances in these fields, we constructed the concept of animal agency, which we define as the ability of animals to actively influence conservation and management outcomes through their adaptive, context-specific, and complex behaviors that are predicated on their sentience, individuality, lived experiences, cognition, sociality, and cultures in ways that shape and reshape shared human–wildlife cultures, spaces, and histories. Conservation practices, such as compassionate conservation, convivial conservation, and ecological justice, incorporate facets of animal agency. Animal agency can be incorporated in conservation problem-solving by assessing the ways in which agency contributes to species' survival and by encouraging more adaptive and collaborative decision-making among human and nonhuman stakeholders.

KEYWORDS

conservation in the Anthropocene, human–wildlife interaction, human–wildlife conflict, human–wildlife coexistence, animal geographies, animal legal theory, animal behavior, interdisciplinary conservation

Resumen: Aunque las prácticas de gestión y conservación de fauna (GCF) han partido históricamente de una gama amplia de áreas académicas, los practicantes se han visto lentos para participar en las conversaciones emergentes sobre los animales como seres complejos, cuya individualidad y sociabilidad influyen sobre sus relaciones con los humanos. Proponemos un reconocimiento explícito de los animales no humanos silvestres como participantes activos en la GCF. Para esto, examinamos 190 estudios sobre las intervenciones y los resultados de GCF para resaltar tres supuestos comunes que respaldan a muchas estrategias actuales de GCF: el comportamiento animal es rígido y homogéneo, la fauna exhibe un comportamiento silvestre idealizado y prefiere hábitats prístinos, y las relaciones humano-fauna son de importancia marginal o secundaria en relación con las interacciones no humanas. Descubrimos que estas intervenciones de gestión no consideran lo suficientemente el aprendizaje, toma de decisiones, individualidad, sociabilidad y relaciones con los humanos de los animales, por lo que llevan a resultados imprevistos y perjudiciales. Para lidiar con estas limitaciones, sintetizamos los avances teóricos que han tenido las ciencias dedicadas al comportamiento animal, la geografía animal y la teoría legal animal que pueden ayudar a los profesionales de la conservación a reformular el concepto de animal y sus relaciones con los humanos. Con base en los avances en estas áreas construimos el concepto de agencia animal, el cual definimos como la habilidad que tienen los animales para influir activamente sobre la conservación y los resultados de manejo por medio de su comportamiento adaptativo, complejo y específico al contexto, los cuales están basados en su sensibilidad, individualidad, experiencias vividas, conocimiento, sociabilidad y culturas, de manera que construyen y reconstruyen las culturas, espacios e historias humano-fauna. Las prácticas de conservación, como la conservación compasiva, la conservación acogedora y la justicia ecológica, incorporan facetas de la agencia animal. La agencia animal puede incorporarse en la solución de los problemas de conservación al evaluar las formas en las que la agencia contribuye a la supervivencia de la especie y al alentar una toma de decisiones más adaptativa y colaborativa entre los actores humanos y los no humanos.

PALABRAS CLAVE

coexistencia humano-fauna, comportamiento animal, conflicto humano-fauna, conservación en el Antropoceno, conservación interdisciplinaria, geografía animal, interacción humano-fauna, teoría legal animal

野生动物保护和管理中的动物能动性

【摘要】野生动物保护和管理的实践历来来自于各种学术领域, 但动物作为复杂生命体, 其个性和社会性影响着它们与人类的关系, 因此实践者很难跟上关于动物不断涌现的讨论。我们建议应明确承认野生非人类动物是野生动物保护和管理的积极参与者。我们调查了关于野生动物保护和管理的干预和结果的190项研究, 并指出目前许多野生动物保护和管理方法的三个常见假设:动物行为是刻板和同质的;野生动物表现出理想化的野生行为, 喜欢原始的栖息地;相比于动物与非人类的互动, 人类与野生动物的关系是边缘或次要的。我们发现这些管理干预措施没有充分考虑到动物的学习、决策、个性、社会性以及与人类的关系, 引起了意想不到的有害结果。为了解决这些缺陷, 我们综合了动物行为科学、动物地理学和动物法律理论方面的理论进展, 这些知识有助于保护专家重新认识动物及其与人类的关系。这些学科深入研究了动物的知觉、适应性、个性、集体决策以及对人类共享环境的参与。基于这些领域的进展, 我们构建了动物能动性的概念, 定义为动物通过其适应性、特定环境和复杂的行为积极影响保护和管理结果的能力, 这些行为是建立在它们的知觉、个性、生活经验、认知、社会性和文化之上的, 其方式塑造和重塑了人类与野生动物共有的文化、空间和历史。保护实践, 如同情心保护、和谐性保护和生态正义, 都包含了动物能动性的各个层面。通过评估动物能动性对物种生存的贡献, 以及鼓励人类和非人类利益相关者之间更多的适应性和合作性决策, 可以将动物能动性纳入到保护问题的解决方案之中。**【翻译: 胡怡思; 审校 : 聂永刚】**

关键词: 人类世的保护, 人与野生动物互动, 人与野生动物冲突, 人与野生动物共存, 动物地理学, 动物法律理论, 动物行为, 跨学科保护

INTRODUCTION

In the face of unprecedented transformations to the biosphere, wildlife conservation and management (WCM) must constantly evolve. We define *WCM* as the practice and study of wildlife conservation, management, and human–wildlife interactions that intersect with the broad fields of human–animal studies. (We use *wildlife* and *wild animals* interchangeably to refer to nonhuman animals that live somewhat autonomously from humans, are self-sufficient, and possess the freedom to reproduce.) WCM draws from diverse disciplines to accomplish its goals of protecting and preserving wild animals to ensure their survival and well-being while considering the well-being of humans who share landscapes with them. Many WCM interventions—from international policies, such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), to national protected areas, to local restoration—are notable successes that can credit their formulation and effectiveness to theories and concepts absorbed from, for example, population dynamics and ecological modeling, monitoring and evaluation, applied statistics, genetics, and geospatial sciences. The practice of WCM has broadened to include humanities and social sciences (Manfredo, 1989; Moon et al., 2019), engage with various forms of expertise and values (Lawrence, 2010; Tengö et al., 2014), and consider possibilities for coexistence between humans and wildlife in human-dominated landscapes (Carter & Linnell, 2016; Frank et al., 2019; Hodgson et al, 2020; Pooley et al., 2017; Woodroffe et al., 2005).

Yet, WCM practitioners have been slow to adopt key findings from disciplines that engage with the complexity of animals' lives and behaviors, their relationships with each other and with humans, and the ways in which these relationships shape the world humans share with wildlife. Despite the emergence of novel WCM approaches that challenge anthropocentric perspectives (e.g., Brakes et al., 2021; Büscher & Fletcher, 2019; Celermajer et al., 2020; Wallach et al., 2020; Washington et al., 2018) and the celebration of animal personhood by respected conservationists (e.g., Jane Goodall) and in the public imagination (Manfredo et al., 2020), many contemporary WCM policies and practices are still based on assumptions that wild animals respond passively to reconfigurations of complex human systems, without considering their influence in shaping these systems.

We considered explicit acknowledgment of wildlife as active participants in WCM. We did so by surveying recent work in the fields of animal behavioral sciences, animal geographies, and animal legal theory. Though emerging from distinct theoretical and epistemological backgrounds, researchers in these fields share an interest in understanding the complexity of animals, their relations to their environments and to humans, and how these dynamics can and should shape human treatment of nonhuman animals. By highlighting convergences of these fields toward similar sensitivities to animals and human–animal relationships, we examined the implications of considering animal agency an integral part of developing nuanced and effective approaches to the practice of WCM. We define *animal agency* in WCM as the ability of animals to actively influence conservation and management outcomes through their adaptive, context-specific, and complex behaviors that are predicated on their sentience, individuality, lived experiences, cognition, sociality, and cultures in ways that shape and reshape shared human–wildlife cultures, spaces, and histories. The modalities of practice presented here have been part of global communities for centuries because many non-Western traditions attribute agency to animals (Hornborg, 2015; Watts, 2013). However, this holistic definition of agency remains underexplored in the majority of current WCM practices.

We critically evaluated conceptual assumptions that underpin dominant forms of WCM and illustrate the potential for enriching views of animals to improve WCM outcomes. For instance, animal agency shares some influences and positions with compassionate conservation (Wallach et al., 2020), convivial conservation (Büscher & Fletcher, 2019), and ecological justice (Kopnina & Washington, 2020), which grapple with the intrinsic value and personhood of all sentient beings and humans' ethical obligations to them. We argue that animal agency can offer a useful lens to understand the successes, challenges, and spaces for growth in novel and established approaches. In doing so, we aimed to complement and support scholarly work reimagining more just and effective WCM futures (Ampumuza & Driessen, 2021; Batavia et al., 2020; Bhattacharyya & Slocombe, 2017; Toncheva & Fletcher, 2021).

WHY INTERVENTIONS PRODUCE UNEXPECTED OUTCOMES

Understanding how WCM activities have unintended outcomes for wild animals and humans can help illuminate shortcomings and address future challenges emerging from increased and novel human–wildlife interactions. Table 1 provides a sample of scenarios in which WCM practices produced unanticipated results. We built the table by reviewing, compiling, and synthesizing 190 peer-reviewed publications in which WCM interventions were evaluated. We conducted targeted searches for commonly used intervention methods (e.g., translocation, reintroduction, fencing). Although not a comprehensive list, it is indicative of the diversity of species, practices, and outcomes associated with mainstream WCM. We augmented this review with authors' prior research involving interactions between humans and leopards (*Panthera pardus*), white-tailed deer (*Odocoileus virginianus*), thesus macaques (*Macaca mulatta*), and wolves (*Canis lupus*). Drawing on the examples in Table 1, we identified 3 common assumptions about animals that unite the examples (full citations to Table 1 sources are in Appendix S1): animal behaviors are rigid and homogeneous, wildlife exhibit idealized wild behavior and prefer pristine habitats, and human–wildlife relationships are of marginal or secondary importance to the goal of species preservation. These assumptions are not held by all conservationists or applied in all management scenarios but, nonetheless, represent pervasive ideas in WCM across species, contexts, and time.

Animal behaviors are rigid and homogeneous (Assumption 1)

Many WCM strategies are based on the assumption that a species or individuals' behavior in one context will remain largely unchanged in another and that individuals of the same species behave uniformly (Table 1). Yet, animals, including wolves, coyotes (*Canis latrans*), elephants (*Elephas maximus* and *Loxodonta* spp.), and leopards, frequently exhibit plasticity of behavior unanticipated by WCM interventions. Testing a predictive habitat suitability model for wolves in Wisconsin developed by Mladenoff et al. (1995), Mech (2006) demonstrated that the model was a "poor predictor of wolf re-colonizing locations in Wisconsin, apparently because it *failed to consider the adaptability* of wolves. Such models should be used cautiously in wolf-management or restoration plans" (Mech, 2006, p. 874) (emphasis added). Yet such models often underlie wolf management strategies (e.g., Mech, 2015; Michigan Department of Natural Resources, 1997; Wisconsin Department of Natural Resources, 1999).

Relatedly, many WCM efforts are predicated on the assumption that interventions will not fundamentally reshape

TABLE 1 Selected wildlife conservation and management interventions, their intended goals, and actual outcomes

Species, management method	Expected management or conservation outcome	Actual outcome of intervention	References[a]	Assumptions[b]
African elephant (*Loxodonta africana*), lethal control, multiple locations	Reduction in crop raiding once so-called problem animal is killed. Human communities will be appeased as problem animals are controlled.	Number of raiders did not decrease because other individuals replaced removed raiders. Problem animal misidentified. Continued community hostility toward elephants and conservation efforts.	Hoare, 2001, 2012	A1
African elephant and Asian elephant (*Elephas maximus*), translocation, multiple locations	Reduction in crop raiding once so-called problem animal is translocated. Elephants will stay at release site and will not occupy or reoccupy new or original sites. Elephants will thrive and cease to break fences if moved to native habitat.	Animals tried to return to their home range. Stressed individuals show PTSD symptoms. Increased mortality. New conflicts around release site. Fence breaking escalated in original location and spread to new location.	Evans & Adams, 2018; Fernando et al., 2012; Hoare, 2001; Pinter-Wollman, 2009; Shaffer et al., 2019	A1, A2
Asian elephant, nonlethal deterrents	Elephants will avoid threatening sounds/spotlights.	Elephants developed tolerance for deterrents and returned to area.	Shaffer et al., 2019	A1
African elephant in south Asia, exclusion through fencing, multiple locations	Elephants will remain outside of fences and not enter human spaces.	Elephants returned to human spaces. Elephants crossed and broke fences. Fences funneled high number of elephants creating conflict with surrounding communities.	Hoare, 2012	A1
African elephant, detusking, Kenya	Once detusked, elephants will not break fences.	Fence breaking reduced but detusked elephants developed new techniques to break fences.	Mutinda et al., 2014	A1
Kangaroo rat (*Dipodomys stephensi*), translocation, USA	Translocations of individuals to newly restored areas will reestablish populations.	Translocations ignored established neighborhood relationships resulting in low reproduction and survival rates.	Greggor et al., 2016	A1
Rhesus macaque (*Macaca mulatta*), translocation, India	Translocation from cities to rural areas will provide less disturbed habitat and reduce or remove nuisance macaque population from urban sites.	Individuals quickly colonized nearest human settlements. Individuals continued to behave aggressively toward humans and native macaques. Site of capture (urban New Delhi) was repopulated Residents split over morality of intervention.	Govindrajan, 2015; Kumar et al., 2013	A1, A2, A3
Coyote (*Canis latrans*), nonlethal deterrents, USA	Use of plastic collars around the necks of sheep will reduce number of attacks on sheep and other domesticated animals.	Coyote adapted their attack behavior to the hindquarters of the sheep.	Blackwell et al., 2016	A1
European badger (*Meles meles*), culling, UK	Culling will reduce the reservoir of TB infection in wild badgers (considered the underlying source of increased infection rates across species).	Cull survivors explored unoccupied territories and deposit infected feces in new locations, contributing to disease spread.	Cassidy, 2012; MacDonald, 2016	A1
Brown bear (*Ursus arctos*), hunting, USA	Hunting will control bear population.	Longer maternal care and potentially slower reproduction to avoid hunting exposure.	Van de Walle et al., 2018	A1

(Continues)

TABLE 1 (Continued)

Species, management method	Expected management or conservation outcome	Actual outcome of intervention	References[a]	Assumptions[b]
Black bear (*Ursus americanus*), translocation, Canada	Translocation will reduce interactions between humans and habituated bears in residential areas.	Low survival of translocated bears. Bears returned frequently to site of capture.	Landriault et al., 2006	A1, A2
Wolf (*Canis lupus*), translocation, USA	Translocation will reduce predation on livestock and encounters with humans.	Most translocated wolves left release areas and traveled to or through areas of livestock production.	Fritts et al., 1984	A2
Wolf, culling, USA	Culling will reduce predation of wolves on livestock and conflicts with humans.	Predation increased as culling led to social disruption and fragmentation of packs and less efficient hunting.	Borg et al., 2015; Brainerd et al., 2008; Fernández-Gil et al., 2016	A1
Wolf, reintroduction, Europe	Wolves will colonize areas of low human population density across Europe.	Spontaneous rewilding in more densely populated areas.	Drenthen, 2016	A1, A2
White-tailed deer (*Odocoileus virginianus*), hunting, USA	Hunting will control overabundant deer population.	Deer ranges shifted away from roads during the hunting season, avoiding areas of greater human activity. Altered deer behavior during hunting season affected endangered Florida panther.	Kilgo et al., 1998	A1, A2, A3
Leopard (*Panthera pardus*), translocation, India	Translocation from peri-urban areas to core of protected areas will reduce leopard population density and minimize attacks and encounters with humans.	Individuals traveled long distances to return to original range. Social disruption at sites of capture and release. Increased attacks on humans.	Athreya et al., 2011	A1, A2
Bengal tiger (*Panthera tigris tigris*), reintroduction, India	Reintroduce tiger population in undisturbed protected areas.	Tiger reintroduction displaced leopards into human-dominated environments, increasing conflicts with humans.	Mondal, 2012	A1
Bottlenose dolphin (*Tursiops truncatus*), reintroduction from captivity, USA	Dolphins will thrive in the wild and supplement endangered or threatened populations, or reestablish a population in former range.	Many individuals did not survive.	Wells et al., 1998	A1, A2
Horses (*Equus ferus*), culling, USA	Removal of feral horses will help restore native habitat.	No removal of horses after years long conflict between Ozark residents and National Park Service. Conflicts between groups emerged from differences in representation of and attachment to horses.	Rikoon, 2006	A3
Multiple species, community displacement, multiple locations	Displacement of human communities from protected areas will reduce detrimental anthropogenic impacts.	Absence of critical anthropogenic activities resulting in loss of landscape and species diversity. Loss of indirect monitoring leading to encroachment by more destructive actors. Increased pressure on natural resources at sites of settlement. Negative attitudes toward conservation.	Fabricius & de Wet, 2002; Cernea & Schmidt-Soltau, 2003; Rangarajan & Shahabuddin, 2006	A3
Beluga whale (*Delphinapterus leucas*), hunting quotas, Canadian Arctic	Imposition of quotas on beluga whale hunting and tightening of hunting restrictions will maintain robust stocks of beluga.	Hunting above quota in response to restrictions seen as unfair, scientifically unsound, rigid, and ignorant of Inuit perceptions of beluga sentience.	Tyrrell, 2007, 2008	A3

(Continues)

TABLE 1 (Continued)

Species, management method	Expected management or conservation outcome	Actual outcome of intervention	References[a]	Assumptions[b]
		Criminalization of subsistence hunting with detrimental cultural, economic, and nutritional impacts for the Nunavik Inuit—threatening the survival of Inuit culture and relationship with beluga.		
		Other important factors for the decline of beluga populations were underexplored (e.g., disease, pollution, loss of habitat, net entanglement).		

[a]Complete references are in Appendix S1.
[b]The 3 common assumptions made in wildlife conservation and management: A1, animal behaviors are rigid and homogeneous; A2, wildlife exhibit idealized wild behavior and prefer pristine habitats; and A3, human–wildlife relationships are of marginal or of secondary importance to other ecological relationships.

animal decision-making (Swaisgood, 2010). This assumption can undermine reintroduction efforts of captive-bred individuals because captivity profoundly influences behavior and decision-making and therefore survival rates in reintroduction programs (Jule et al., 2008). Further, disruption of the social fabric of animal communities by culling, translocation, and reintroduction can impair the survival and longevity of targeted species (Teixeira et al., 2007). For example, culling elephants can lead to the breakdown of social systems among the affected population, driving the emergence and spread of hyperaggressive behaviors (Bradshaw et al., 2005).

Wildlife exhibit idealized wild behavior and prefer pristine habitats (Assumption 2)

Much WCM is based on the assumption that animals will return to an idealized state of wildness if offered appropriate environments. Translocation and reintroduction of leopards, macaques, elephants, and dolphins illustrate this point (Table 1). Emerging from the assumption that wild animals inherently prefer undisturbed or pristine habitats and that these preferences are fixed (Osko et al., 2004), habitat preference is understood to be directly correlated with habitat quality, which itself is assumed to have a direct relationship with the level of human disturbance or population density. This relationship is used in wildlife population models that underlie many WCM decisions (Battin, 2004). However, real-world habitat preferences and resource selection among individual animals contradict outcomes predicted by these population models (Nielsen et al., 2002; Osko et al., 2004), as do the results of numerous wildlife reintroduction and translocation programs where animals attempt to return to disturbed sites.

Human–wildlife relationships are of marginal or secondary importance (Assumption 3)

Although human dimensions are recognized and integrated in WCM scholarship and practice (Bennett et al., 2017; Manfredo, 1989; Treves et al., 2006), WCM efforts often fail to consider differences in the ways human individuals, communities, and cultures view and value animals. Most WCM approaches are based on species' biological, ecological, or economic value, thus classifying them as overabundant, invasive, endangered, game, and so forth. However, to many people wildlife are sentient beings, kin, deities, or community members (e.g., Borish et al., 2021; Nair et al., 2021; Tyrrell, 2008). By narrowly considering human–animal relationships, WCM practice often overlooks traditions that engage with wildlife as unique individuals or cultural entities distinct from but related to humans—discounting the shared histories, geographies, and dependencies that create these relationships.

The exclusion of complex human-wildlife relationships from WCM results in unanticipated outcomes that run counter to the objectives of contemporary conservation—to protect threatened species with the support of local communities. For example, the failed removal of feral horses in the Ozark Riverways (USA), intended to restore native ecosystems, was partly due to a disregard for the horses' local historical, cultural, and emotional significance (Rikoon, 2006). In the Canadian Arctic, Inuit cultures consider beluga whales (*Delphinapterus leucas*) sentient beings that are deeply connected to communal practices. However, state-sponsored plans that included rigid quotas on whale hunting ignored Inuit knowledge of and relationships with whales. This engendered a breakdown of Inuit livelihoods and cultural identity and a distrust of comanagement, contributing to decisions to exceed state-imposed hunting quotas (Tyrrell, 2007, 2008).

ACKNOWLEDGING THE ROLE OF ANIMAL AGENCY

Our review of common assumptions and unanticipated outcomes highlighted key shortcomings in conceptualizations of wildlife in WCM that come from a shared historical lineage. Western scientific thought, heavily influenced by Judeo-Christian views of human dominion over nature and Descartes' treaty on animals in the 16th century, has a long history of treating animals as automata (Crist, 2013). Animals are considered inferior and subordinate to humans, lacking emotion, free will, self-consciousness, or personhood. Although societies across space and time—ranging from the European middle ages to contemporary world religions—have acknowledged animal sentience and laboratory studies increasingly demonstrate personality and empathy in animals, the cartesian perspective has carried through to contemporary Western conceptualizations of wildlife. Current WCM approaches perpetuate the idea that humans can control and contain animals. Often when wildlife leave designated spaces or exhibit novel behaviors, they are viewed as overabundant, out-of-place, or problematic. As a result, these approaches often devalue habitats that are not seen as pristine and strive to excise behaviors outside of those observed in idealized conditions. These practices routinely exclude communities that consider animals sentient (Berkes, 2012). In doing so, WCM efforts can delegitimize relationships and spaces characterized by more complex human–animal engagements (Blaser, 2009; Borish et al., 2021) and dismiss forms of knowledge about animals that are not deemed scientific (Saberwal, 2000).

Disciplinary overview

We selectively reviewed recent scholarship in 3 fields that reconceptualize animals and their relationships with humans: animal behavior, animal geographies, and animal legal theory. These fields question many of the premises of contemporary WCM discussed above. We did not conduct a comprehensive literature review; rather, we sought to highlight concepts that could enrich WCM.

Animal behavioral sciences explore why animals act the way they do through studies of expression, intelligence, learning abilities, culture, sociability, cognition, and the range and flexibility of these characteristics. Throughout the 20th century, behaviorism—which considered behavior strictly a response to stimuli—strongly influenced the study of animal psychology. Some branches of behavioral sciences have since taken a more comprehensive view of behavior and its drivers—understood to be influenced by personality, temperament, experience, mood, attitudes, social context, and so forth (Levitis et al., 2009). For instance, the field of cognitive ethology focuses on the study of animal intelligence and demonstrates that animals' thoughts, feelings, and social systems are more developed than previously thought (Bekoff, 2002; de Waal, 1989). Drawing from Darwin's theory that the difference between animals and humans is in degree, not kind, cognitive ethologists engage with "all ways in which animals take in information about the world through the senses, process, retain and decide to act on it" (Shettleworth, 2001, p. 278). While originating in higher primate studies, animal ethological research has broadened to species ranging from ants to cetaceans (Brakes et al., 2021; MacDonald & Ritvo, 2016; Shettleworth, 2010). These studies contribute to rejecting the static view of animals as passively occupying existing environments (Barua & Sinha, 2017).

Animal geographies have emerged as a rich and heterogeneous subdiscipline (Buller, 2014) to respond to the "deafening silence about nonhumans" in social theory (Wolch & Emel, 1995, p. 632). Building on methodologies and frameworks from geography (including actor–network theory, posthumanist, feminist, Marxist, Indigenous, and cultural geographies), animal geographers also draw from diverse animal-centric fields, including animal ecology and behavioral sciences (Barua & Sinha, 2017; Lorimer & Srinivasan, 2013; Wolch & Emel, 1995). Animal geographers are interested in the multiple ways animals intersect with human societies (Urbanik, 2012) and complicate mainstream views of animals. They reject utilitarian representations of animals as objects and resources under human control with no influence on human lives. By exploring the various temporal, spatial, and place-based relationships among humans and animals, animal geographers consider the geographies of animals, their active participation in the construction of landscapes (Wilbert & Philo, 2000), and their heterogeneous, fluid, intertwined subjectivities (Govindrajan, 2018; Holloway, 2007). They critically examine the ways in which dominant discourses on animals are rooted in capitalist traditions that commodify nonhumans and devalue their relationships to humans (Wolch & Emel, 1995). The literature explores human relationships with companion animals (Haraway, 2008), farmed animal welfare (Miele, 2011), and wild species (Ampumuza & Driessen, 2021; de Silva & Srinivasan, 2019; Dempsey, 2010; Toncheva & Fletcher, 2021). In the context of WCM, animal geographers consider wild animals political actors engaged in WCM through their relationships with humans and other species (e.g., Boonman-Berson et al., 2016; Evans & Adams, 2018).

Animal legal theory has its roots in animal philosophy, which long ago established that animals possess sufficiently similar mental and emotional capacities to those of humans that they should be given similar moral consideration (Jamieson, 2018; Regan, 1983; Singer, 1975). Foundational work in animal legal theory argues for inherent rights for all organisms and questions the Western legal status of animals as objects, solely the property of humans (Francione, 1995; Stone, 1972). Drawing from animal ethics and political animal philosophy, animal legal theorists view animals as sentient beings with moral standing, subjective experiences, and abilities to shape their own and others' lives. Therefore, they possess individual and collective interests that should be represented in human institutions and included in decision-making regarding the governance of spaces (Celermajer et al., 2020; Cochrane, 2018; Garner, 2013; O'Sullivan, 2011; Regan, 1983; Rowlands, 1997; Wise, 2000). Animal legal theory has moved from a conceptual academic interest to a field with tangible consequences for and benefits to animals through its shaping of outcomes in legislation and litigation (e.g., Cohen, 2006; Dunn & Rosengard, 2017). Much of the theory and

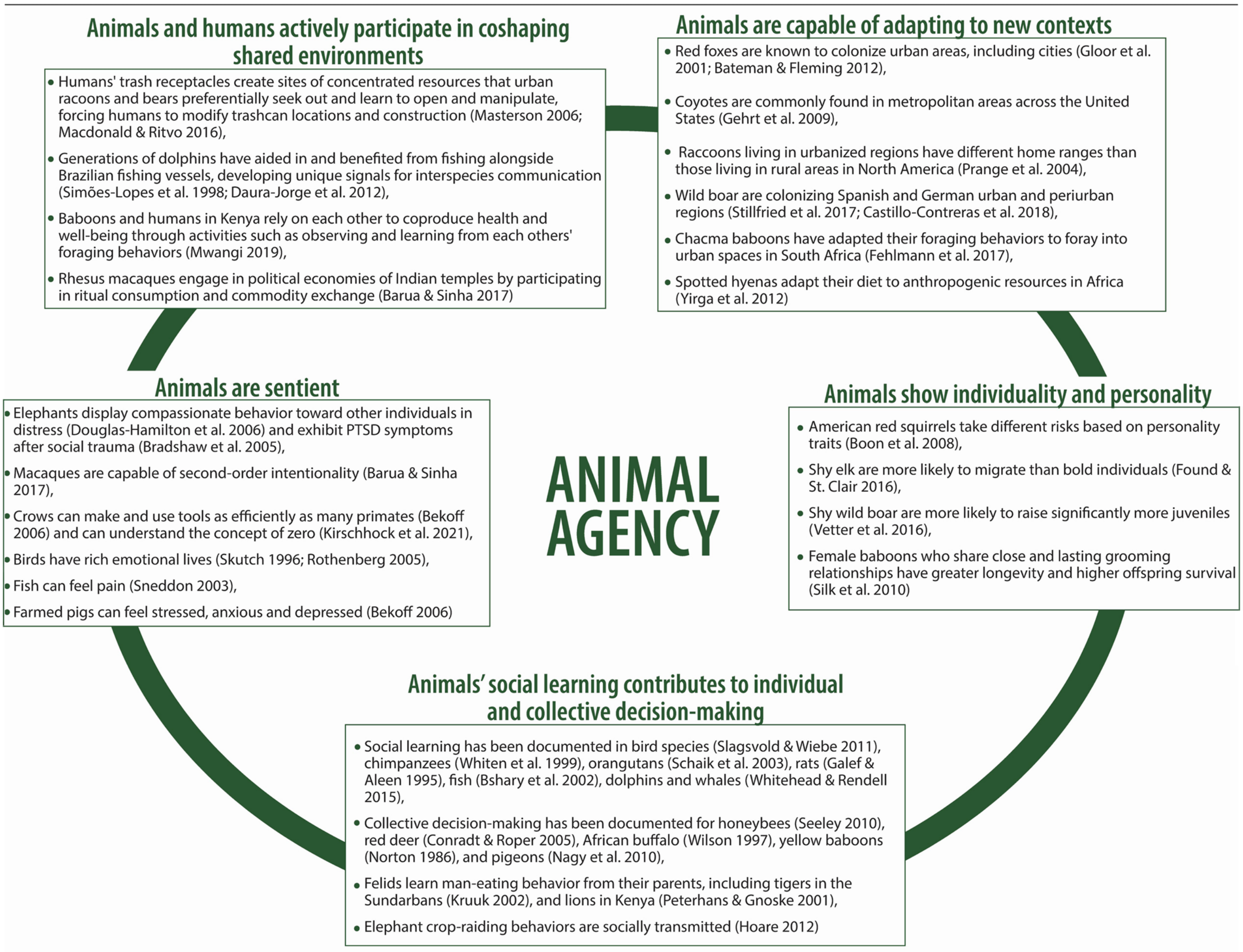

FIGURE 1 Selected examples that illustrate important components of animal agency
Full references are in Appendix S1.

practice of animal law exists on a spectrum from animal welfare on one end (concerning the responsibilities of humans toward protecting animals' best interests) to animal rights on the other (a deontological position that animal interests are inherent and inviolable, which legal systems should be designed to defend, as is the case with human rights). Some animal legal philosophers go further by asserting that animals have agency—they are not only aware of their surroundings and interactions, but also proactively shape them—a concept animal law and philosophy as a whole have been slow to embrace (Jamieson, 2018).

KEY LESSONS FROM DIVERSE DISCIPLINES

The collective findings from these fields challenge assumptions that underpin many mainstream WCM approaches. We considered 5 conceptual contributions that have implications for the ways animals are protected, managed, and treated in WCM. Figure 1 summarizes selected case studies that illustrate each of these concepts. Full citations to sources in Figure 1 are in Appendix S1.

First, animals are sentient. That is, they have feelings and intelligence. Many species possess a shared sense of morality, empathy, and justice (Bekoff & Pierce, 2017). Numerous behavioral studies of bird and mammal species identify expressions of empathy and emotions, including fear, pain, and distress (Masson & McCarthy, 2016). Animals are also reflective and capable of "remembering the past and planning for the future" (Kaplan, 2016, p. 201).

Second, animals are capable of adapting to new contexts. Their behavioral plasticity allows animals to adapt and habituate to different conditions. Animal behaviorists, geographers, and legal theorists demonstrate that animals can modify behaviors when faced with change, including human disturbance (Griffin et al., 2017), by drawing on past experiences and interests (Donaldson & Kymlicka, 2016; Gullo et al., 1998; Hodgetts & Lorimer, 2015). Generalist species adapt to anthropogenic changes by finding novel ways to exploit resources in human-dominated landscapes (Devictor et al., 2008; Figure 1). Experiences can also be learned and transmitted over generations (Berger, 2008).

Third, animals show individuality and personality. Individuals from the same population can have personality traits that set them apart from others. Behavioral scientists have established that in most studied species, individuals exhibit idiosyncratic behavioral differences (Blackwell et al., 2016; Dall & Griffith, 2014; Merrick & Koprowski, 2017; Réale et al., 2010). Both genetic and nongenetic factors drive these differences (Honda et al., 2018; Réale et al., 2007) and influence the decisions individuals make (Réale et al., 2010).

Fourth, animals' lived experiences and social learning contribute to individual and collective decision-making. All 3 disciplines provide strong evidence for the sociality of animals, which allows them to develop distinct languages (Bekoff, 2002) and the capacity for collective decision-making. Social behavior varies across time and space producing communication idioms and cultures (Bekoff, 2002; de Waal, 1999; Laland & Janik, 2006). There is growing evidence that animal culture, defined as "information or behavior—shared within a community—which is acquired from conspecifics through some form of social learning" (Whitehead & Rendell, 2015, p. 12), exists in a wide range of wild animals (Brakes et al., 2021).

Finally, animals and humans actively participate in coshaping shared environments. Animal geographers and animal legal theorists understand human–animal interactions as a product of complex relational processes in which humans and animals are active participants. Both fields recognize animals' influence in shaping the natural world—as agents of ecological processes—but also in coshaping humans' socioeconomic, cultural, and political worlds (Dempsey, 2010; Hobson, 2007). For example, female bottlenose dolphins (*Tursiops truncatus*) have had lasting and complex relationships with fishers in Brazil, and individual dolphins have socially learned cooperative foraging tactics that benefit both dolphins and humans (Bezamat et al., 2020; Simões-Lopes et al., 2016). Rhesus macaques (*Macaca mulatta*) participate in the political economies of Indian temples by engaging in ritual consumption and commodity exchange with humans (Barua & Sinha, 2017). Beluga whales, polar bears (*Ursus maritimus*), and caribou (*Rangifer tarandus*) are enmeshed in the cultural and socioeconomic lives of many communities in the Arctic, and over centuries their relationships with humans have endured and developed (Borish et al., 2021; Kishigami, 2005; Tyrrell, 2007). Elephants in Sri Lanka are "companion species" because they have coevolved with people over millennia such that "their genetics, anatomies, behaviors, feelings, social groupings, and wider ecologies all bear a human signature. At the same time, the language, culture, religions, agriculture, and economies of their human coinhabitants carry a pachyderm trace" (Lorimer, 2015, p. 23). Even the activities of slugs (multiple species) in domestic gardens shape fine-scale geographies and humans' relationships with their shared environments (Ginn, 2014).

Animals have been considered guardians, deities, companions, rivals, nations, community members, and coconspirators that contribute in direct and indirect ways to the survival of both human cultures and wild species (Bhattacharyya & Slocombe, 2017; Blaser, 2009; Lorimer, 2015; Nadasdy, 2007; Nair et al., 2021). These views acknowledge animals' intentions, emotions, and cultures that they share with humans in a common social, spiritual, and ecological world (Umeek-Atleo, 2011). Although often associated with non-Western, precolonial traditions (Berkes et al., 2000), meaningful relationships with animals that engage with their personhood and shared culture also exist throughout Western societies, often in vastly divergent ways (e.g., hunters [Kelly & Rule, 2013], and in animal rights advocates [Rudy, 2011]).

Defining animal agency for WCM

Our review draws attention to the need for greater recognition of wild animals' complexity and intentions in their interactions with humans in WCM contexts. The concept of animal agency captures this complexity. The term *agency* can be broadly understood as "the capacity to produce a phenomenon or modify a state of affairs" (Jepson et al., 2011, p. 230). Although the term agency is used differently across disciplines (Carter & Charles, 2013; Jamieson, 2018; Jepson et al., 2011; Nash, 2005; Steward, 2009; Teubner, 2006), we integrate findings from the 3 fields to build a definition of *animal agency* in WCM contexts that encompasses the complexities discussed above.

We defined *animal agency* as the ability of animals to actively influence WCM outcomes through their adaptive, context-specific, and complex behaviors that are predicated on their sentience, individuality, lived experiences, cognition, sociality, and cultures in ways that shape and reshape shared human–wildlife cultures, spaces, and histories. Adopting animal agency as a lens in WCM helps practice move beyond perceptions of wildlife as manipulable objects, recognizes animals' active participation in WCM, and gives valence to worldviews that have long incorporated dimensions of animal agency in their engagements with the environment.

INTEGRATING ANIMAL AGENCY INTO WCM SCIENCE AND PRACTICE

Barriers exist to incorporating agency in WCM practice. First, incorporating facets of animal agency into predictive models of animal behavior is challenging (Budaev et al., 2019). Quantitative, automata-based methods are well established, scalable, parsimonious, and inexpensive (Budaev et al., 2019) and therefore commonly used in WCM planning. In contrast, acknowledging animal agency introduces nonuniformity, uncertainty, and complexity at the modeling, planning, and implementation stages. Integrating agency into predictive models can require more complex, expensive, and computationally intensive simulations (Budaev et al., 2019). Second, although many practitioners implicitly recognize animal agency (e.g., Boonman-Berson et al., 2016), there are structural and institutional challenges to widespread application in WCM—such as the difficulty in updating established systems of practice and policy and the entrenchment of cartesian approaches by those in power (Jacobson & Decker, 2006). Despite these challenges, facets of animal agency are already integrated and can be further explored within existing and emergent WCM practices.

Incorporating animal agency in conservation prioritization

Conservationists closely consider metrics that treat animals primarily as quantifiable stock when defining conservation priorities and measuring success (e.g., viability, endemism, population size, genetic diversity [Brakes et al., 2019]). However, WCM efforts that only consider tangible and measurable components of animal life at the expense of less tangible, more plastic aspects (e.g., behavioral traits, cultural diversity) ignore essential characteristics of individuals, groups, and ecosystems that contribute to survival.

Integrating animal agency into WCM strategies can help identify and conserve agentic qualities essential for species' survival (Berger-Tal et al., 2016; Blumstein & Fernández-Juricic, 2010; Greggor et al., 2016; Smith & Blumstein, 2013). Applied conservation behavior research has expanded to explicitly consider how individuality, personality, and learning produce heterogeneous responses across individuals and their implications for ecological and population-scale processes (Brakes et al., 2021; Merrick & Koprowski, 2017). Personalities of animals can influence metrics as fundamental as population estimates. For example, individuals that are less perturbed by human presence are more likely to be counted (Biro, 2013). Bold and exploratory individuals tend to exhibit greater tolerance for noise, human activity, and other forms of disturbance. They are more likely to make use of conservation infrastructures (e.g., nest boxes, artificial habitats, etc.), come into conflict with humans, transmit and acquire zoonotic diseases, and colonize new areas (Found & St. Clair, 2016; Greggor et al., 2016; Honda et al., 2018; Merrick & Koprowski, 2017). Coupling data on personality and behavioral traits associated with habituation to humans and disturbance tolerance with population and genetic diversity data can help identify vulnerable, isolated populations (Riley et al., 2014). Similarly, incorporating learning and behavioral diversity into landscape connectivity and dispersal modeling has serious implications for conservation corridor planning because models have strikingly different results when different behavioral characteristics are included (Elliot et al., 2014). By inquiring how individuals, groups, or populations engage with and respond to landscapes, an agency-based approach illuminates how animals shape contexts to meet their needs under different scenarios, potentially altering conservation outcomes. For example, different populations of brown bears (*Ursus arctos*) have attuned behaviors toward humans based on the varying degrees of protection across Bulgarian regions (Toncheva & Fletcher, 2021). Coupling agency-based framings that consider wildlife's behavioral plasticity and decision-making with well-established practices to understand spatial patterns—such as tracking with GPS collars, wildlife cameras, and satellite images—is also worth greater exploration to identify vulnerable individuals.

Animal agency can also enrich WCM priorities, including the preservation of animal social systems and culture (Brakes et al., 2019, 2021; Marzluff & Swift, 2017), as advocated for in animal culture conservation approaches (Laiolo & Jovani, 2006). Culture can affect crucial survival skills that contribute to the persistence of social groups and potentially whole populations. For example, accounting for dolphins' and wolves' social systems was key to successful reintroduction programs (Ferguson, 1996; Milstein, 1995; Wells et al., 1998). African elephant matriarchs (*Loxodonta africana*) accumulate knowledge regarding their social and ecological environment, transmitting information crucial to group survival (Mccomb et al., 2001). Yet traditional approaches prioritize younger individuals' reproductive potential (Brakes et al., 2019). These studies demonstrate the importance of not only protecting genetic diversity and reproductive capacity, but also cultural and social systems for species survival. These ideas are gaining traction in conservation science (Griffin et al., 2000; MacDonald, 2016) and informing human–wildlife conflict management (Brakes et al., 2021; Greggor et al., 2017; Marzluff & Swift, 2017)—such as identifying culturally significant units in the protection of small and endangered populations (e.g., Ryan, 2006; Whitehead et al., 2004). Further, practitioners and conservation institutions recognize the importance of cultural traits at the individual and group levels and at the population and species levels (Brakes et al., 2021). For example, the Convention on the Conservation of Migratory Species of Wild Animals is exploring the implications of conserving cultural traits, such as clan culture among sperm whales (*Physeter macrocephalus*) and nut-cracking culture in banded mongooses (*Mungos mungo*) for the preservation of these species (CMS, 2017, 2018).

Managing wild animals with their agency in mind

Viewing animals as active participants allows one to reconsider how conservationists and wildlife managers can engage in WCM. Practical WCM experiments already include various facets of animal agency (although different terminology might be used); results suggest avenues forward for animal-agency-centered WCM.

Linking cognitive science, animal cognition, and evolutionary ecology, an increasing number of behavioral ecologists incorporate animal personalities, life histories, emotions, learning abilities, and motivations to better model animal adaptive decision-making (Budaev et al., 2019). For example, in traditional husbandry, carcasses of animals killed by predators or accidents are generally quickly removed. However, this may actually limit the ability of domesticated animals to learn about predators and the importance of avoiding dangerous areas (Marzluff & Swift, 2017).

Animal decision-making is also emphasized in the kincentric ecology approach (Bhattacharyya & Slocombe, 2017) that foregrounds multispecies collaborative management in shared socioenvironmental systems. To manage human–seagull conflict, the city of Leiden in the Netherlands experimented with gull–human collaboration in negotiating nesting locations that met both gull (*Larus argentatus*) and human needs (Meijer, 2016). In doing so, managers acknowledged the role gulls can play in management efforts through "interspecies decision-making" (Meijer, 2016, p. 64). Other examples of humans leveraging the participation of animals in conservation include the involvement of beavers (*Castor canadensis*) in watershed management (Woelfle-Erskine & Sarna, 2013) and captive Asian elephants in mitigating human–animal conflicts (Münster, 2016). These examples illustrate the possibilities that emerge by considering animals as agents of territorialization able to occupy human-modified environments and as creative participants in adaptive experimentation.

Other contemporary management strategies are noteworthy for their consideration of animal agency within more traditional paradigms. Using deterrence mechanisms (strikes), a wildlife management approach in Colorado was intended to teach black bears (*Ursus americanus*) to avoid human spaces. Although this initiative was based on the assumption of uniform bear behavior, managers whose role was to implement strikes often bent the rules and used their intimate knowledge of bear autonomy and individuality to decide which animals to target (Boonman-Berson et al., 2016). This example demonstrates the need to experiment with context-specific, adaptive strategies that leverage existing, carefully nurtured human–animal relationships (Boonman-Berson et al., 2016). It also speaks to recent discussions regarding the moral implications of making decisions in WCM. For example, Batavia et al. (2020) argue for considering the concept of moral residue in WCM, recognizing the ethical challenges of WCM and encouraging conservationists to sit with the emotional dimension of their missions. Finally, this example illustrates that many managers know that animals have agency and implicitly acknowledge it by integrating it into management practice, despite institutional norms and discourses that discount its importance.

Further, viewing animals as active participants in conservation policy-making raises pertinent questions of whose knowledge is valuable in understanding and representing animals' perspectives, interests, and rights (Toncheva & Fletcher, 2021). Considering agency encourages us to more closely examine worldviews that have been perceived as lacking scientific rigor, but are products of decades or centuries of integration between human and nonhuman lives. Many management practices are rooted in human–wildlife reciprocal relationships and derived from multigenerational experience-based knowledge (e.g., Kideghesho, 2009; Mukul et al., 2012; Toncheva & Fletcher, 2021). For instance, Rayne et al. (2020) show how Indigenous knowledge systems in Aotearoa New Zealand can improve outcomes of conservation efforts, such as the translocation of understudied species. In the Canadian Arctic, Inuit hunters' knowledge of muskoxen and caribou life histories, population dynamics, and body conditions was crucial to conserve these species (Tomaselli et al., 2018). In Bulgaria, experience-based knowledge of local communities (and especially hunters) is key to cohabitation with bears. There, humans and bears have developed relations of mutual "trust" and "respect" through repeated, nonconflictual, peaceful encounters (Toncheva & Fletcher, 2021). This type of knowledge can enrich conservation policy-making and potentially inform the appointment of human "trustees" to advocate for animal rights in WCM efforts (Cochrane, 2018). In the same vein as Etuaptmumk (Mi'kmaw for "Two-Eyed Seeing" [Bartlett et al., 2012]), which advocates for the coexistence of various knowledge paradigms, we argue that animal agency can support the development of hybrid deductive and inductive reasoning and address complex issues with all available and critical sources of information necessary to face the ongoing loss of global biodiversity. Further, recognizing the many ways of being with and viewing animals is necessary to avoid trivializing or alienating communities directly affected by WCM interventions. This is particularly important in the case of Indigenous Peoples who have been marginalized through centuries of colonial conservation approaches, resulting in the loss of shared human–animal worlds. Although it is crucial to be attentive to the ways local knowledge can be misunderstood, simplified, or instrumentalized, centering animal agency in conservation practices can contribute to efforts that respect and recognize the approaches of Indigenous Peoples—who currently manage or have tenure rights over one-quarter of the world's land surface, representing about 40% of the world's terrestrial protected areas (Artelle et al., 2019; Garnett, 2018).

Finally, integrating animal agency into conservation allows more nuanced discussions of, and can potentially augment, existing and emergent practices. WCM will always be an endeavor held in tension by different goals, worldviews, and ontologies of what is worth conserving and how to conserve it. Engaging with animal agency will not remove the challenge of balancing different views or easily solve ecologically, politically, and culturally fraught conservation challenges that inherently involve trade-offs (see, for example, Oommen et al. [2019] and their critique of compassionate conservation). The degree to which each facet of the animal-agency concept needs to be engaged may vary among species, ecological systems, and local contexts. For these reasons, we argue that considering animal agency can draw attention to and spur conversations about fundamental questions and tensions that often go unspoken in mainstream WCM. Driving questions may include: How will humans and wildlife engage with and affect different WCM efforts? Is there room for WCM plans to adapt as diverse humans and animals learn from each other? How can plans incorporate more than the biological value of a species? Are the human communities most closely engaged with animals able to contribute and increase their knowledge and expertise under this management regime? How can their relationships be honored, maintained, and supported? What animal cultural traits and relationships does this make room for, and what does it inhibit? How will these interventions produce new interspecies relationships, cultures, and politics? We encourage managers and stakeholders interested in exploring the ramifications of an animal-agency lens to ask these questions within contexts described in Table 1.

These questions have relevance regardless of whether managers use mainstream WCM approaches or emergent practices and can help WCM practitioners evaluate plans, develop scenarios, engage with other stakeholders, make room for surprises, and imagine multiple futures. We thus present animal agency as a concept with the potential to connect wildlife, Indigenous and local communities, scholars, conservationists, and wildlife managers to enhance context-specific and adaptive WCM practice. These approaches have the potential to create spaces for better collaboration, inclusion, and well-being for both animals and humans.

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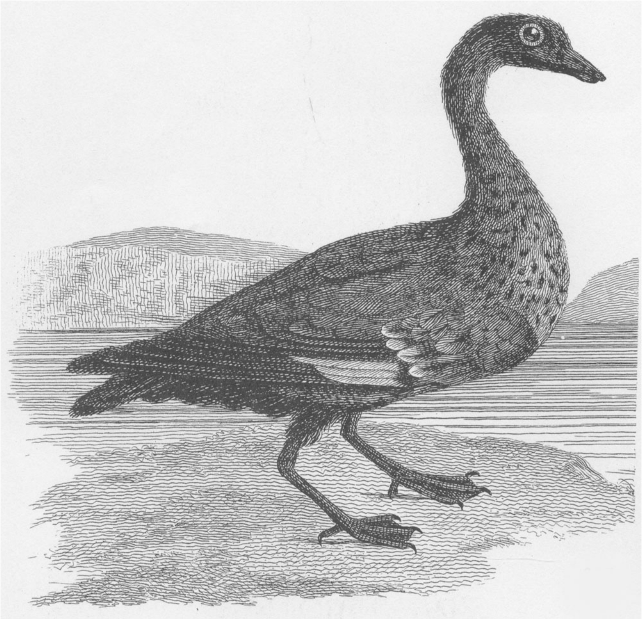

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Urban biodiversity management using evolutionary tools

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Cities are fully functioning ecosystems and are home to no-analogue communities of species that interact with each other and which are subject to novel urban stressors. As such, biodiversity can evolve in response to these new urban conditions, making urban species a moving target for conservation and management efforts. An evolving urban biodiversity necessitates integrating evolutionary insights into management for these efforts to be successful in a dynamic urban milieu. Here we present a framework for categorizing urban biodiversity from a management perspective. We then discuss a suite of example management tools and their potential evolutionary implications—both their opportunities for and potential consequence to management. Urban ecosystems are proliferating but, far from being ecological lost causes, they may provide unique insights and opportunities for biodiversity conservation. Determining how to achieve urban biodiversity priorities while managing pest species requires evolutionary thinking.

Urbanization is intensifying and expanding worldwide. Human-dominated urban environments—once considered ecological 'lost causes'—are functioning ecosystems and are increasingly recognized as valuable targets for species conservation and biodiversity management[1–4]. Recent research has demonstrated that the species inhabiting cities are capable of rapidly changing in response to anthropogenic environments[5–7]. For example, the urban heat island effect and urban pollutants have both been implicated in driving animals to rapidly evolve adaptations to urban life[8–12]. Additionally, some aspects of plant reproductive biology have also been demonstrated to evolve in the urban environment[13,14]. The realization that species are capable of quickly evolving in response to urban contexts has driven a flurry of academic and popular interest in urban evolution in recent years[7,15–19].

Evolutionary insight offers an untapped opportunity to better manage urban biodiversity but also highlights the fact that biodiversity is a moving target in the complex, dynamic urban milieu. Human activities drive the fastest rates of evolutionary change, and those human effects are most pronounced in urban environments[5,20,21]. At the same time, there is an increasing realization that urban areas are important targets for biodiversity conservation and for engaging communities typically underrepresented in the sciences with ecology and evolutionary biology[1–4,22]. Therefore, conservation, evolutionary biology, and community engagement uniquely overlap in urban ecosystems. Given the burgeoning research focus on urban evolutionary biology, the time is right to consider how evolutionary insights can refine urban biodiversity management and conservation efforts.

Conservation and management decisions in urban landscapes can and must account for evolutionary processes. Failure to do so is likely to hinder biodiversity management efforts or result in unintended consequences that include, but are not limited to, target and non-target species declines, wasted funds and labour, or benefits to harmful or pest species (Fig. 1)[21,23]. Here we present a framework for categorizing urban biodiversity from a management perspective. We then discuss a suite of example management tools and their potential evolutionary implications, both their opportunities for and potential consequences to management. Our aim here is to bridge the gap between conservation practice and evolutionary biology in urban environments. We hope that this framework can serve as a resource for practitioners and academics to integrate evolutionary considerations into common management scenarios and to guide future research efforts so as to maximize actionable insights for urban biodiversity management.

Urban communities

Urbanization is both destructive and constructive. In the process of building our cities and suburbs—including constructing green and blue infrastructure or habitats—people alter or eliminate existing habitats and replace them with novel environments. Urban biological communities are largely unintentional assemblages of the native species that persisted during urbanization and the native and non-native species that have since colonized or been introduced (Box 1). This results in unique no-analogue ecological communities: species assemblages that have never existed elsewhere in space or time. Similarly, no-analogue communities resulting from climate change are predicted to produce novel ecological and evolutionary scenarios, complicating conservation and management decisions in protected areas[24,25]. In urban settings, this difficulty in predicting eco–evolutionary outcomes is further exacerbated by the myriad novel stressors and human dimensions inherent in urban ecosystems.

Populations of management target species are nested within broader urban communities; the populations comprising these communities may evolve both in response to different urban pressures as well as to each other. Thus, management decisions affecting one target species will likely cascade to affect other species in the community, but these cascading effects can be difficult to anticipate. Therefore, consideration of the evolutionary history of urban species assemblages (Box 2) and the evolutionary trajectory of urban biodiversity targeted for conservation and management is a research priority. Understanding the evolutionary principles undergirding these interactions and outcomes is the first step in designing effective urban management strategies.

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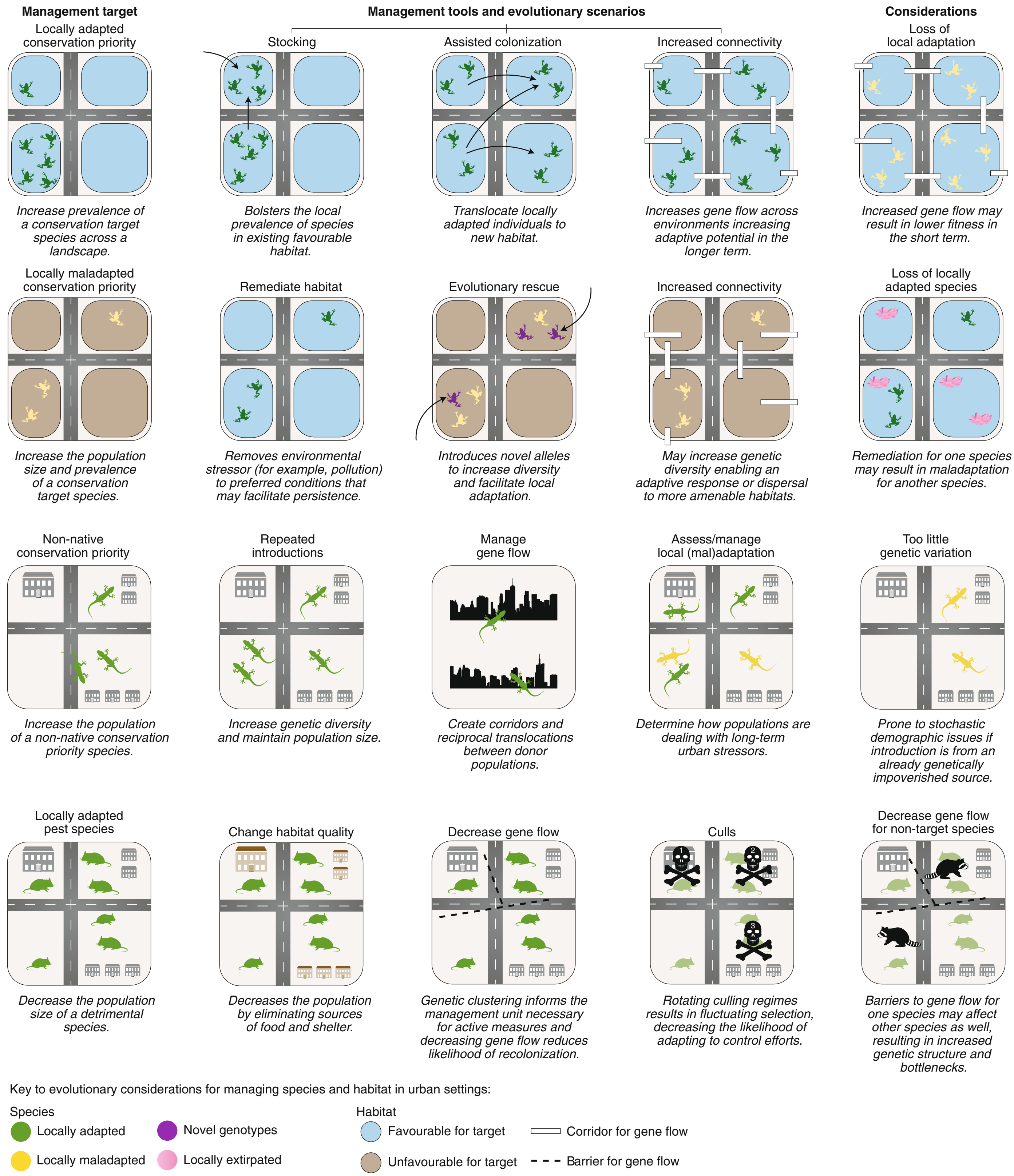

Fig. 1 | Illustrating potential management tools in multiple evolutionary scenarios for four management targets. The four management targets are: conservation priority species that are either locally adapted or maladapted, a non-native conservation priority species, and a locally-adapted pest species. Evolutionary insight can guide management decisions that either directly manage populations (for example, various forms of translocations) and/or habitat (for example, enhancing corridors or barriers to gene flow). Initiating an urban management plan can also have evolutionary impacts on other species in the urban community that warrant considerations.

Box 1 | Urban no-analogue communities

Urban biological communities are comprised of at least five key categories of organisms that may or may not be the focus of active conservation measures, but whose evolutionary potential may be affected by urban processes and management efforts (Fig. 1). Some species (for example, coyotes or racoons) may fall within more than one category depending on the context.

Native

Desirable, not conservation priorities: we care about these species (for example, acorn ants) because we appreciate native biodiversity and recognize their roles in biological community and ecosystem processes. Studying evolutionary processes can be informative to better manage urban ecosystems to favour these species over ecologically similar but non-native species.

Conservation priority: these are species (for example, western swamp turtle) that are generally of conservation concern and whose ranges might be entirely relegated to urban environments[4]. It is critical to determine the evolutionary processes affecting populations of conservation priority species to tailor management for adaptive states or adaptive processes.

Introduced

Neutral: these species (for example, pigeons or clover) are common and typically tolerated. Because we recognize that these species probably play important roles, studying the evolution of these urban species is necessary for understanding the eco–evolutionary processes influencing their roles in urban communities and ecosystem processes.

Harmful: these species (for example, rats) represent threats to human health and property as well as conservation-priority biodiversity. Often our management goals are to minimize the impact and persistence of these species. Understanding harmful species' adaptive states and processes could help us manipulate habitats and populations to minimize their fitness and aid eradication efforts.

Exotic but endangered: Shaffer[2] proposed a novel management option ('urban biodiversity arks') where biologists purposefully introduce and sustain endangered species in urban habitats outside their native ranges. The composition of urban biological communities is often unplanned, however, we have a potential conservation opportunity to encourage species of concern (for example, red-crowned amazon parrots in Los Angeles).

Evaluating urban evolution

Evolutionary terminology and concepts (Box 2) are sometimes used colloquially but have more restrictive scientific definitions, sometimes causing confusion. For our purposes here, we have highlighted key evolutionary concepts in two broad research approaches: trait-based studies and population genetics. We note, however, that other areas of evolutionary research like phylogenetics—the study of evolutionary relationships among species—are informative to conservation but are beyond the scope of our discussion here.

Trait-based approaches. Populations living in urban ecosystems often differ from rural populations in morphology (for example, body size or limb and head dimensions), physiology (for example, stress hormones, metabolism and pollution susceptibility), behaviours (for example, phenology, attraction to light, and boldness), reproductive traits (for example, gamete type or quality), and others[5,16,26]. These traits may vary among urban subpopulations or between urban and non-urban populations as a result of evolution or phenotypic plasticity (Boxes 2 and 3).

Evolution or plasticity? Trait differences have evolved when those differences are genetically based and heritably passed on to subsequent generations. In contrast, phenotypic plasticity can give rise to trait variation when different environmental conditions alter the expression of the same genotype—an individual organism's particular genetic makeup—in different contexts (Box 3). Unlike evolved trait differences, phenotypically plastic trait variation is generally not heritable because only the environment, not the genetic underpinnings, differs among populations.

These pathways for trait variation are not mutually exclusive: trait variation among populations can be the product of both evolved differences and phenotypic plasticity. One particularly interesting research frontier is investigating the extent to which the capacity of a trait to be phenotypically plastic may itself evolve in urban environments. This scenario has been elegantly demonstrated using ants responding to urban heat islands. Acorn ants, regardless of whether they are from urban or rural populations, can tolerate higher temperatures if raised in warmer rather than cooler environments (a plastic response), but urban ant populations exhibit elevated tolerance of extreme urban heat (an evolved difference) and an increased plasticity in coping with rapid changes in temperatures (evolved plasticity)[11,12,27].

Is the trait variation adaptive? Observed trait variation in populations across urban and rural gradients may or may not confer a benefit in the urban environment. A population's trait expression is considered adaptive if it increases the fitness (higher survival, more offspring) of organisms in that particular ecological context. Trait expression may also be maladaptive, reducing the success of organisms in a particular environment. Or, trait variation may simply be neutral and have minimal positive or negative consequences. Critically, both evolved and plastic trait differences can be adaptive, maladaptive or neutral[28,29]. Moreover, continuing environmental change, including future urban land-use decisions and climate change, could change the adaptive, maladaptive or neutral consequences of an urban population's trait expression, ultimately rendering urban biodiversity management a moving target. Phenotypic variation, whether it be genetic or plastic, can have negative or deleterious effects to individuals.

Observed trait variation—whether due to evolution or to phenotypic plasticity—is sometimes presumed to be adaptive in urban areas[30]. However, we strongly caution against this assumption as, untested, it can lead to overestimation of a species' adaptive capacity and, correspondingly, the impact of conservation and management actions on that species. It remains one of the foremost challenges for urban evolutionary biology research to conclusively determine the plastic or evolved mechanisms driving trait differences in urban populations, but from a management perspective, the need to conclusively distinguish between plastic and evolutionary processes is critical.

Population genetics approaches. In contrast to trait-based research, population genetics studies focus on understanding the genetic composition of and variations in gene / allele frequencies among populations. By the numbers, most urban evolutionary studies to date have focused on population genetics[5,7,31,32]. Most urban population genetics studies focus on quantifying neutral evolutionary processes including, for example, population genetic structure, genetic drift, gene flow, inbreeding, genetic diversity and so on (Box 2)[31]. While neutral processes, by definition, do not demonstrate adaptive evolution, insight from population genetics studies has important implications for managing urban biodiversity.

Box 2 | Evolutionary concepts for urban biodiversity

Adaptive: when a trait conveys a fitness advantage in a particular habitat. Both evolved trait differences (that is, adaptations) and plastic trait changes can be adaptive.

Maladaptive: when a trait reduces fitness in a particular habitat. Both evolved trait differences (that is, maladaptations) and plastic trait changes can be maladaptive.

Local adaptation: higher fitness of local, genetically based phenotypes over foreign ones due to different selection pressures. An urban population could be locally adapted to the city if urban individuals display higher survival/fitness when reared in the city compared to non-urban individuals transplanted into the city.

Local maladaptation: higher fitness of foreign, genetically based phenotypes over local ones. An urban population could be locally maladapted to the city if urban individuals display lower survival/fitness when reared in the city compared to non-urban individuals transplanted into the city.

Absolute maladaptation: when a population's mean fitness is lower than its replacement rate, ultimately leading to population decline.

Relative maladaptation: when a population's mean fitness is lower than another population's mean fitness. An urban population may be relatively maladapted compared to a non-urban population but its mean fitness may still be equal to or above its rate of replacement, suggesting a relatively stable population.

Phenotypic plasticity: non-genetically based phenotypic variation (Box 3). Plastic trait variation could be adaptive, maladaptive or neutral. Myriad studies show phenotypic differences between urban and non-urban environments; it remains an open question as to whether these are genetically based differences or phenotypic plasticity.

Developmental plasticity: when a trait's expression is the result of developmental variation among individuals due to being reared in different environmental conditions, rather than genetic differences. Developmental plasticity is typically irreversible.

Phenotypic flexibility: when a trait's expression can vary throughout an individual's lifetime due to experiencing different environments or changes in a single environment. Flexible phenotypes are reversible.

Gene flow: the movement of genes or alleles (DNA variants of a given genetic region or locus) among populations. Gene flow by itself does not directly reflect dispersal because dispersed individuals may not breed in their new population and therefore do not contribute to gene flow. In urban environments, various landscape features may facilitate gene flow (for example, green spaces, sewers, subway tunnels and road crossings), while others act as barriers (for example, roads and buildings). Gene flow may be endogenously regulated by a given species' natural history or exogenously influenced by anthropogenic actions like translocations or habitat modification.

Genetic drift: changes in allele frequencies within a population due to random chance in survival or reproduction. This differs from natural selection which reflects differential survival and reproduction between different genotypes in a given environment. When there is little or no gene flow among different populations, random processes will produce changes in allele frequencies between populations by chance; these changes do not represent adaptation to each populations' particular environment.

Urban experimental design: two approaches commonly used in assessing urban evolution are common garden and reciprocal translocation experiments. Common garden experiments take individuals from different populations and rear them in a single environmental condition to assess trait heritability and phenotypic plasticity. Reciprocal transplants involve rearing alternate populations in opposing environmental conditions to assess relative adaptation. For example, rearing urban and rural populations from embryos or eggs in the lab could examine whether an urban trait occurs due to plasticity, evolution, or both. Transplanting the populations (from rural to urban, and from urban to rural) would provide insight on whether any observed trait variability connoted a fitness advantage in either setting. Common garden experiments can, however, demonstrate fitness benefits if organisms are exposed to an isolated urban stressor like pollution or extreme heat as part of the experiment.

Inbreeding / outbreeding depression: inbreeding depression occurs when reproduction between closely related individuals results in reduced genetic diversity and increased susceptibility to stochastic processes and future environmental change. The opposite of this is outbreeding depression which occurs when reproduction between more distantly related individuals increases genetic diversity and reduces a population's mean fitness (local adaptation) to its current environmental context.

Population genetic structure: genetic variation across time and/or space reflective of dispersal and population boundaries due to physical or behavioural barriers. Population genetics assesses differences in the frequency of alleles that fluctuate within and across populations due to various forces like natural selection, mutation, gene flow and genetic drift. Population genetic structure can reflect ecological processes and evolutionary history and can influence a population's evolutionary potential and trajectory. In urban environments, population structuring may occur at relatively coarse scales (between urban and non-urban environments) and also at relatively fine scales (within an urban landscape) due to different barriers to movement and natural selection to different local pressures.

Translocation: a management strategy aimed at moving individuals of a species from one habitat or population to different habitats or populations. Translocations should be informed by source and destination populations' evolutionary histories as well as possible evolutionary consequences. A translocation could take several forms, including stocking to enhance population sizes or genetic diversity of a smaller population, colonizing empty but putatively suitable habitat, or evolutionary rescue, whereby individuals from different populations are introduced to increase genetic diversity to enhance a population's adaptive processes.

For instance, populations with higher genetic diversity should be better able to adapt to future environmental changes overall, including in cities[33]. Therefore, if an urban population of conservation concern was shown to have high rates of inbreeding and low genetic diversity, management efforts may prioritize introducing new individuals into the population to help bolster that population's adaptive potential against urban stressors. Additionally, understanding rates and directions of gene flow illustrates the extent to which popula-

Box 3 | Plastic phenotypes in urban environments

Here, we focus on two forms of phenotypic plasticity: developmental plasticity and phenotypic flexibility[41,42]. We note that other forms of phenotypic plasticity exist, for example, epigenetic inheritance[43,44], but these forms are difficult to identify and are beyond the scope of our discussion here.

Distinguishing between developmental plasticity and phenotypic flexibility is important for conservation. In particular, developmental plasticity is typically not reversible. This means, for example, that translocating older individuals from a rural population to an urban population may be problematic if the phenotype expressed by that individual is better suited for rural environmental conditions. If developing in urban conditions results in an individual presenting a phenotype that is better suited for those specific conditions, then translocations into various urban habitats may be more effective if they target embryos or young offspring to maximize the chances that a plastically developed trait is best matched to local urban conditions. Phenotypic flexibility, on the other hand, can be reversible and so may facilitate moving organisms at various life stages, including older individuals, if a trait (for example, physiological or behavioural) can adjust to the new conditions at various life stages.

Both forms of plasticity underscore the importance of using common garden and/or reciprocal transplant experiments to determine whether trait differences in urban settings are evolved or plastic (Box 2). Simply measuring trait differences between urban and non-urban populations is necessary but insufficient to determine whether those differences arise from plasticity or evolution, let alone whether that trait variation is adaptive, maladaptive or neutral.

Discerning whether an organism's urban trait arises due to environmentally driven plasticity or evolution is critical to management. If traits enabling a conservation-priority species to persist in the urban environment are due to phenotypic plasticity, conservation practitioners perhaps have a greater number of management tools at their disposal to facilitate the spread and maintenance of that species or to enhance habitat connectivity among urban populations (Fig. 1). However, if this species persists instead because it is locally adapted to urban stressors, then practitioners may be left with fewer management options (Fig. 1).

tions are connected. Gene flow can be a population's 'double-edged sword': on the one hand, gene flow can help a population maintain genetic diversity, but gene flow may also introduce an influx of alleles that are poorly suited for a particular environment (see Case study 1)[23]. Understanding the genetic structure of urban populations is an important step in any conservation and management plan because this information can elucidate biologically relevant management units within the city (see Case study 2)[34–36].

A review of 167 urban population genetics studies found that urbanization nearly always affects population genetics parameters, but the effects on population genetics are inconsistent across species and cities[31]. While urbanization is often predicted to inhibit gene flow, isolate populations and reduce genetic diversity, this is not always the case. Roughly one-third of studies suggest genetic diversity of urban populations is enhanced by urbanization, in contrast to predictions. For example, urban features like roads or subways can sometimes impede and other times increase gene flow, depending on the species. Because urbanization does not have a single clear population genetics consequence, it is critical for practitioners to account for the biology of target species and the heterogeneity in local urban environments that might dictate gene flow, genetic drift or population genetic structure before engaging in management activities.

Understanding within- and among-city population genetics can provide crucial context for how urbanization structures biodiversity and facilitates or impedes the movement and success of individuals. Integrating this area of research into management plans will likely enhance the precision and success of urban biodiversity conservation. It is important to note that population genetics and trait-based research are not mutually exclusive. On the contrary, mutual insight from both areas of research will likely provide the most informative guidance for managing urban biodiversity.

Managing with evolutionary tools

Making an urban management plan should begin with delineating goals identifying one or more target species, should follow with compiling all available information on evolutionary history (genetic diversity, gene flow, population structure, local adaptation, phenotypic plasticity and so on), and can then be informed by considerations of the intended and unintended evolutionary consequences for target and non-target species. Actively managing urban biodiversity can involve many approaches[4,37,38], including, for example, remediating low-quality habitat, adding or protecting putatively suitable habitat, or enhancing connectivity among populations either through habitat modifications (for example, under/overpasses or corridors) or via translocations (Fig. 1).

These tools broadly aim to directly manage habitats or populations, with consequences for a species' adaptive state and its adaptive processes (sensu Derry et al.[23]). Managing for an adaptive state means maximizing a population's current fitness and, as a consequence, minimizing phenotypic and genotypic variation. Such an approach may increase fitness in the short term but minimize a population's evolutionary capacity for adapting to future environmental change. Managing the adaptive process of a population means enhancing genetic diversity to promote longer-term evolutionary potential. Managing for the adaptive process may result in maladaptation to current local urban conditions in the short term, but so long as the population is not absolutely maladapted (Box 2), this may be relatively inconsequential for longer-term persistence.

Managing an evolving urban biodiversity

One goal of urban management is to bolster existing populations and increase the prevalence of native species of conservation concern or management interest across an urban landscape. If the target species are locally adapted to urban conditions, reducing gene flow from non-urban populations or from different types of urban habitats may be helpful to maintain a particular locally-adapted state. However, doing so may come at the cost of minimizing genetic diversity in urban populations, thereby limiting potential adaptation to future environmental change, urban or otherwise.

Conversely, enhancing gene flow for a locally-adapted population might bolster genetic diversity, enhancing the population's adaptive potential, but with the consequence of reducing its current adaptive state (Fig. 1; locally adapted conservation priority). If the target population is not locally adapted, but is generally plastic to environmental conditions, enhancing gene flow either through translocations or increasing habitat connectivity among urban subpopulations and between urban and non-urban populations may be beneficial for enhancing the urban population's adaptive potential[39]. Alternatively, locally maladapted conservation priority species (Fig. 1; *Locally Maladapted Conservation Priority*) may require mitigating urban stressors to minimize the degree of maladaptation and/or increasing gene flow through population manipulations – such as translocating individuals from other populations – or habitat modification.

Case study 1 | Pollution and urban frogs

Consider amphibians inhabiting urban stormwater ponds. These ponds are designed to collect stormwater runoff from urban surfaces, and, as such, concentrate myriad contaminants that are harmful to amphibians[45–48]. Nevertheless, research shows that a number of amphibian species—including species of conservation concern—use these ponds regularly, often as readily as they use natural ponds[45,49–52]. One common goal of urban habitat management is to increase green space and connectivity among populations. Yet, in the case of urban stormwater ponds, increasing connective greenspace between ponds may connect a population of a conservation priority amphibian species that is locally adapted to the chemical pollutants in its pond to a rural population or different urban subpopulation that is not adapted to the pollution. The greenspace could thus unintentionally introduce maladapted genes into the pollution-adapted urban amphibian population, to its detriment. However, if amphibians in these communities exhibit plastic responses to pollutants, then increasing connectivity among urban amphibian populations will likely have little impact on pollution susceptibility.

Case study 2 | Native and introduced pests

In black widow spiders—a venomous native pest in the southwestern United States—within-city genetic diversity is higher than diversity in rural areas, gene flow and population connectivity is higher among populations within cities, and population genetic differentiation is lower within cities relative to rural areas[53]. Perhaps most importantly, particularly urban subpopulations of black widows act as highly connected hubs that facilitate the spread of urban individuals[54]. In addition to common population and landscape genomic methods, such a network analysis approach may be useful for identifying regions of the city that are central to a pest's persistence and dispersal. While black widow movement may be facilitated generally in cities, identifying and targeting management towards these key hubs may minimize the spread of these pests. For introduced pests, like brown rats, population genetics research has demonstrated substantial within-city variation in population structure, genetic diversity and gene flow[34–36]. This work identifies management units that can be used to focus eradication efforts and identify potential dispersal corridors. Reducing urban habitat quality may also minimize local adaptation, and varying culling techniques across space and time (for example, using different poisons or trapping methods) can also help limit local adaptation in urban populations.

For pest species, managing both adaptive states and adaptive processes are probably key to reducing populations and corresponding detrimental effects (Fig. 1; *Locally Adapted Pest Species*). A pest species that is highly locally adapted to particular local urban conditions may be easier to manage than a pest with extensive phenotypic plasticity. For the first case, it may be possible to change the environment so that population becomes relatively maladapted, but the latter may quickly and flexibly adjust to environmental changes.

Identifying corridors and barriers to gene flow as well as population genetic structure within cities will be important for identifying tractable management units across the urban landscape and for minimizing dispersal that could enhance each subpopulation's adaptive capacity[35,36]. Understanding gene flow is particularly important for determining the best management techniques for pest species.

The species assemblages inhabiting urban ecosystems are largely comprised of the species that persisted during urbanization or have since colonized these environments. Outside of planted species, urban biological communities are largely unplanned species assemblages. One management option to consider would be to "rewild" cities by intentionally reintroducing native species to potentially suitable urban habitats. Doing so would offer an opportunity to experimentally test whether native species that are currently absent from cities are not present because they cannot migrate into built landscapes, because they cannot tolerate urban conditions (e.g. chemical, light or sound pollution) even if they could penetrate cities, or both. For example, experimental work in exurban ponds demonstrated that ponds where wood frogs (*Rana sylvatica*) are currently absent can adequately support these amphibians, suggesting that the developed terrestrial landscape currently limits their colonization of these urbanized ponds[55]. Carefully choosing source populations to rewild cities and suburbs would allow biologists to track the ecology and evolution of those populations that persist. Doing so could enable conservation practitioners to "adaptively" manage urban biodiversity by tracking their changing adaptive states and adaptive processes. To our knowledge, rewilding is not actively employed in urban landscapes – at least not with fauna – but may offer great potential for enhancing native biodiversity in cities.

Shaffer[2] recently proposed taking a rewilding approach one step further, treating cities and suburbs as "urban arks", i.e. spaces that can help to bolster threatened and endangered species *outside* their native ranges. Selecting urban ark species (Fig. 1; *Non-native Conservation Priority*) requires careful consideration about the capacity of these taxa to plastically adjust or rapidly adapt to urban contexts, the likelihood of becoming maladapted (Box 2) to certain urban environments, and if continued introductions or translocations between replicate introduction cities or between urban subpopulations are necessary to maintain adaptive processes.

Considering the consequences

As with most management activities, manipulating populations or habitat for one target population likely reshapes the ecological and evolutionary processes acting on other members of the community. For instance, improving habitat connectivity to increase gene flow for a target species may also create corridors for introduced pest species to invade new urban habitats (Fig. 1; considerations). This increased connectivity may also have the unintended consequence of eliminating genotypes that have become locally adapted to particular local urban stressors. Additionally, habitat management to minimize dispersal of an introduced pest could have consequences for gene flow in a non-target native species (Fig. 1; considerations).

If a population is locally adapted to a particular urban habitat feature and management remediates this urban stressor, are the consequences for the locally adapted population positive, negative or neutral? The answer to this question is likely specific to a target organism's natural history, the particular urban stressor, and mechanism (for example, physiological or behavioural) experiencing natural selection. In urban stormwater ponds, if amphibians are locally adapted to urban contamination (Case study 1), what happens if pollution entering ponds is cleaned up? Are there costs to being evolved to a contaminant that is no longer present (Fig. 1)? Remediating contamination may ultimately have neutral consequences for the target population; while the urban population may be adapted to contamination relative to non-urban populations, its fitness living in polluted urban water may still be reduced relative to living in the absence of the contaminant. Additionally, while this target species may have persisted in the city because it adapted to contamination, other species may have previously been absent in the city because they were unable to adapt to the contamination.

Remediating contamination may facilitate these other species colonizing urban ponds, perhaps increasing competition with the target species of conservation concern. Under such a scenario, adapting pollution tolerance may facilitate a target species' persistence, in part because doing so minimized competition with other species that could not adapt.

Towards urban evolutionary management

How do we integrate evolutionary insight into urban biodiversity management? Ideally, managers would define urban management goals and could explore the evolutionary processes that have and continue to shape their target species. This would include assessing genetic diversity, gene flow (rather than just migration), genetic bottlenecks and population genetic structure. This also includes identifying the urban stressors that could limit their target population's success and experimentally testing the adaptive state of the target population to these stressors. Managers could then formulate plans to manage adaptive states or processes in light of the potential evolutionary consequences. Given limitations on time and money, comprehensive genetic and experimental analyses may not be feasible on management-relevant timescales. Even so, a thought exercise (for example, scenarios in Fig. 1) that considers the potential evolutionary processes shaping biodiversity can help identify which urban management decisions would likely help maximize the management success of target populations given uncertainty in existing adaptive states and processes.

Considering evolutionary processes provides a relatively untapped opportunity to improve urban biodiversity management. Sometimes evolution can be useful, facilitating how we manage species of conservation concern and even pests. In other instances, evolutionary dynamics can make management more challenging. We can use evolutionary insight in our urban management practices but doing so entails accurately understanding and communicating the various evolutionary processes shaping the species living in our cities and suburbs. Evolution is also not a conservation panacea: some species will never have the chance to adapt to urban environments, and conservation practitioners may not have the opportunity to assess the evolutionary biology of target species. Considering evolutionary processes offers new opportunities for maximizing outcomes and minimizing unintended consequences for urban biodiversity management.

Using evolutionary ideas to manage urban biodiversity is no small task. Survey research has identified contrasting familiarity with evolutionary principles between conservation practitioners and evolutionary biologists as one of the biggest barriers to effective, evolution-informed conservation[40]. Similarly, academic scientists are unlikely to have the same degree of practical experience as conservation practitioners in urban planning and managing biodiversity and habitat. Effective evolution-informed urban conservation will require a cross-disciplinary approach integrating expertise from conservation practitioners with evolutionary biologists, ecologists, urban planners, social scientists and geographers.

Managing biodiversity in our cities and suburbs necessitates working on many parcels of private property and in relatively dense human communities. This presents numerous challenges but also exciting opportunities for deputizing neighbours into conservation efforts and for tangibly illustrating evolution unfolding right in their backyards. Urban evolutionary biology has not only become a research interest but it has also captured broad popular interest[18,19]. By drawing on evolutionary insights, we have the opportunity to simultaneously improve urban biodiversity management and engage communities with a richer understanding of the evolutionary rules of life.

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Author contributions

M.R.L. initiated the project. Both authors contributed to the writing and editing of the manuscript.

Competing interests

The authors declare no competing interests.

Additional information

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