Global Ecology and Conservation 22 (2020) e00953

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: http://www.elsevier.com/locate/gecco

Original Research Article

Assessing the impact of regulations on the use and trade of wildlife: An operational framework, with a case study on manta rays

Hollie Booth [a, b, c, *, 1, 3], Simon Pooley [c, d, e, 2, 4], Tom Clements [b, 4], Mochamad Iqbal Herwata Putra [f, 5], W. Peni Lestari [g, 5], Sarah Lewis [f, h, 5], Luke Warwick [b, 5], E.J. Milner-Gulland [a, 4]

[a] The Interdisciplinary Centre for Conservation Science, Department of Zoology, University of Oxford, 11a Mansfield Rd, Oxford, OX1 3SZ, UK
[b] The Wildlife Conservation Society, 2300 Southern Boulevard, Bronx, NY, 104 60, USA
[c] Department of Life Science, Silwood Park Campus, Imperial College London, Buckhurst Rd, Berks, SL5 7PY, UK
[d] Department of Geography, Birkbeck University of London, London, WC1E 7HX, UK
[e] Department Of School of Life Sciences, University of KwaZulu-Natal, Pietermaritzburg, South Africa
[f] Marine Megafauna Research Group, Misool Foundation, Savu Sea Program, Jl. Eltari, Larantuka, 86219, Indonesia
[g] The Wildlife Conservation Society Indonesia Program, Jalan Malabar 1 No.11, Babakan, Bogor Tengah, Kota Bogor, Jawa Barat, 16128, Indonesia
[h] The Manta Trust, Dorchester, Dorset, DT2 0NT, UK

ARTICLE INFO

Article history:
Received 6 September 2019
Received in revised form 20 January 2020
Accepted 1 February 2020

Keywords:
Elasmobranchs
Impact evaluation
Law enforcement
Wildlife trade
Compliance
CITES

ABSTRACT

Overexploitation represents a significant threat to wildlife, with the severest impacts felt by slow growing, economically valuable species. Governments often seek to address this through regulating utilisation and trade of species, which is commonly catalysed by multilateral environmental agreements (MEAs) such as the Convention on the International Trade of Endangered Species (CITES). However, it is often unclear to what degree CITES and associated regulations lead to tangible conservation outcomes. Robust impact assessments are needed to understand whether regulations are effective for achieving biodiversity conservation goals, and to learn lessons for future policy interventions. Yet such assessments are hindered by data paucity, bias, complexity and uncertainty. Here we discuss key challenges for assessing the impact of regulations on the use and trade of wildlife, and offer a practical approach to overcome them. Our approach combines an integrated framework for collating and analysing disparate and methodologically inconsistent data with a robust process to establish causal inference (and hence assess impact). This framework and process can be applied to any regulation, species or country context. To demonstrate its utility we apply this approach to the case of manta ray utilisation and

* Corresponding author. The Interdisciplinary Centre for Conservation Science, Department of Zoology, University of Oxford, 11a Mansfield Rd, Oxford, OX1 3SZ, UK.

E-mail addresses: hbooth@wcs.org, hollie.booth@zoo.ox.ac.uk (H. Booth), s.pooley@bbk.ac.uk (S. Pooley), tclements@wcs.org (T. Clements), iqbalherwata@gmail.com (M.I.H. Putra), plestari@wcs.org (W.P. Lestari), sarah@mantatrust.org (S. Lewis), lwarwick@wcs.org (L. Warwick), ej.milner-gulland@zoo.ox.ac.uk (E.J. Milner-Gulland).

[1] Current Address: The Interdisciplinary Centre for Conservation Science, Department of Zoology, University of Oxford, Oxford, United Kingdom.
[2] Current Address: Department of Geography, Birkbeck University of London, London, United Kingdom.
[3] This author contributed the majority of the work, including conceptualisation, formal analysis, investigation, methodology and writing - original draft preparation.
[4] These authors contributed equally to this work, in conceptualisation, supervision, review and editing and funding acquisition (TC).
[5] These authors contributed equally to this work, in particular: data curation, resources, validation and review and editing.

https://doi.org/10.1016/j.gecco.2020.e00953

trade (*Mobula alfredi* and *M. Birostris*) in Indonesia. This case study is particularly important due to the recent increase in the number of commercially important elasmobranchs listed in CITES appendices, with Parties adopting various national-level regulations to implement their CITES commitments. However, it is unclear to what degree these listings lead to meaningful regulatory reform, and much-needed reductions in fishing mortality, utilisation and onward trade. Indonesia is also a priority country for effective regulatory reform, due to its role as a major source country for international elasmobranch trade. Overall, we highlight challenges and opportunities for assessing the impact of wildlife trade regulations, which are generalisable across species and contexts, and provide the first attempt to assess the impact of such regulatory change on manta ray mortality in a source country. We also offer recommendation for future implementation and evaluation, emphasising the importance of mixed-methods, multiple datasets, and explicit acknowledgement of bias and complexity.

1. Introduction

1.1. Wildlife trade and regulation

Anthropogenic exploitation of wildlife is major threat to biodiversity and ecosystems (Broad et al., 2002; IPBES, 2019). This issue is of increasing concern to policy-makers due to fast-growing legal and illegal markets, which in turn lead to overexploitation of natural resources ('t Sas-Rolfes et al., 2019; IPBES, 2019; May, 2017). Governments may seek to mitigate the threat of overexploitation by regulating the use and trade of species. Such regulatory measures are often catalysed by multilateral environmental agreements (MEAs), such as the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). CITES aims to ensure that international trade in wildlife does not threaten the survival of species in the wild. It can lead to improved species management in wildlife source countries by requiring that Parties develop domestic legislation to implement its requirements with respect to: exporting (from their own country) and importing (into their country), thus regulating international trade in species. With a legally binding framework for action, and associated compliance mechanisms, CITES is considered one of the world's most powerful MEAs (CITES, 2016; Huxley, 2000). While becoming a signatory to CITES is voluntary, a total of 183 countries (95% of all countries) are Parties to the convention. However, the utility and conservation impact of CITES continues to be debated, with critics noting issues surrounding oversimplification of wildlife trade issues, overreliance on regulation, failure to address market forces and cooption by non-state actors (Challender et al., 2015a; Challender and MacMillan, 2019; Cochrane, 2015; Rowan, 2019).

Regulations on use and trade of wildlife come in many forms, depending on their purpose and context, and may act at a range of geographic scales and supply-chain levels. Broadly, wildlife regulations can be classified into three categories: 1) supply-side measures to manage harvesting, 2) transactional measures to manage trade, and 3) demand-side measures to manage consumption ('t Sas-Rolfes et al., 2019). Such regulations may seek to fully prohibit use and trade of a species, or foster sustainable harvesting and use. CITES focuses only on international wildlife trade, though is implemented through domestic measures. For species listed on Appendix I of CITES, international trade is not permitted except in exceptional circumstances. For species listed on Appendix II, trade must be controlled to ensure utilisation is compatible with the survival of the species in the wild (i.e., sustainable). To support this, Parties to CITES are required to conduct Non-Detriment Finding (NDFs) studies to prove that any international trade in Appendix II species is sustainable. Trade may then be regulated through quotas and permits. Countries are not required to regulate domestic markets for CITES-listed species, though may choose to adopt stricter domestic measures than stipulated by CITES, to regulate use and trade of wildlife as per their own national priorities.

This is not a comprehensive review of all different types of regulations. Our point is that wildlife regulations are diverse, and the specific regulation(s) adopted by a country will depend on the species, markets and national and international context. Henceforth we refer to this suite of potential regulations as "wildlife use and trade regulations" (noting that "use" here specifically refers to direct consumptive use).

Regulations are typically actioned in diverse and complex socio-ecological systems. As such, success or failure (i.e., in terms of achieving tangible conservation outcomes) can be dictated by local-level context-specific factors or exogenous macro-economic or environmental factors (Berkes, 2007; Burn et al., 2011; Underwood et al., 2013). Unintended and perverse consequences of wildlife trade regulations are well documented (Challender et al., 2015a). In addition, data on the use and trade of regulated species, and their status in the wild, is often limited, biased or ambiguous ('t Sas-Rolfes et al., 2019; Friedman et al., 2018; Gavin et al., 2010; Robinson and Sinovas, 2018). Together, these issues hinder meaningful monitoring, causal analysis and attribution of the impacts of wildlife use and trade regulations. Therefore, although regulation and enforcement are commonly applied to alleviate the threat of overexploitation to species, the effectiveness of such measures for improving the state of biodiversity continues to be debated (Challender et al., 2015b; Friedman et al., 2018).

In this paper we explore key challenges for assessing the impact of wildlife trade regulations, and offer a practical approach to tackling these challenges. We illustrate our approach through a pertinent case study on national-level regulation of elasmobranch use and trade. Such impact assessments are important for researchers and policy makers, in order to measure progress towards biodiversity conservation goals and learn lessons for future regulatory interventions.

1.2. Regulation of elasmobranch trade

Sharks and rays (collectively, elasmobranchs) are amongst the world's most threatened, under-managed and data deficient species groups (Dulvy et al., 2014a,b; Dulvy et al., 2017). Large elasmobranchs are particularly vulnerable to trade-driven extinction due to their conservative life history characteristics and high value body parts (McClenachan et al., 2012). With growing concern regarding the sustainability of global elasmobranch utilisation and trade (Dulvy et al., 2014a,b; Worm et al., 2013), there has been an increase in efforts to regulate use of these species (Davidson et al., 2016; Friedman et al., 2018; Ward-Paige, 2017).

For elasmobranchs, regulations can encompass complete bans on harvest, trade, or consumption, as well as fisheries management measures such as time-area closures, effort restrictions or retention bans with live release protocols (Booth et al., 2019b, 2019a; Shiffman and Hammerschlag, 2016). CITES has played a major role in driving regulatory changes (Friedman et al., 2018). Thirty-eight commercially important species were listed on Appendix II at the past three Conferences of the Parties (CoPs), bringing the total to 46 species listed on Appendices I and II (38 on Appendix II, 5 on Appendix I (Fig. 1)).

Appendix II listings stipulate that Parties to CITES must ensure international trade in these species is legal and non-detrimental to the survival of the species in the wild. Yet in order to halt population declines, and have meaningful conservation impact for elasmobranchs, CITES listings need to translate into measurable reductions in fishing mortality, utilisation and onward trade. Yet there are concerns from some regarding the applicability of CITES for marine species (Vincent et al., 2014), and the impact of regulation on fishing mortality to date remains questionable (Davidson et al., 2016; Friedman et al., 2018). While expert opinion suggests these CITES listings have positively influenced fisheries governance; fundamental stock-, fisher- and market-related data for demonstrating tangible conservation impacts remain limited (Friedman et al., 2018). This lack of data hampers science-based management of fisheries (Simpfendorfer and Dulvy, 2017), the formulation of appropriate regulations for use and trade, and an assessment of their impacts. With insufficient scientific advice to make management decisions, governments may be slow to act, or may implement inappropriate policies, perpetuating a cycle of under-management and overfishing (Lack and Sant, 2011).

With additional elasmobranch species listed on CITES at the 18th CoP in [August 2019], it is both important and timely to provide practical tools that can support Parties to gather and analyse elasmobranch utilisation and trade data. This can support formulation and evaluation of robust management decisions for elasmobranchs, and track progress towards national and international management goals.

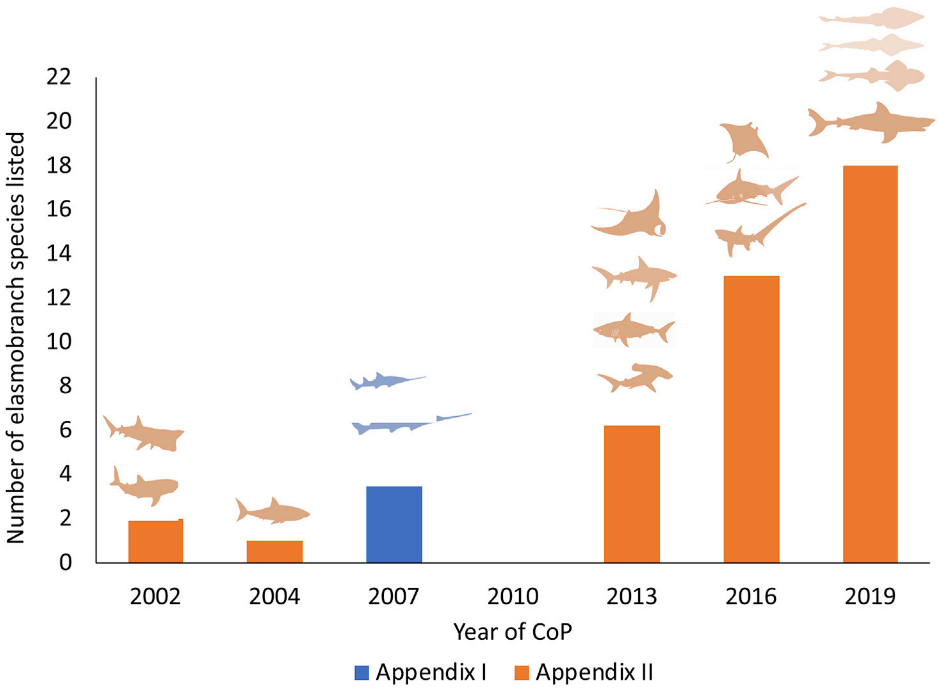

Fig. 1. A history of elasmobranch listings on CITES Appendix I and II. Year denotes the year of the CoP in which the CITES decision was made. This may not be the same year the listing came in to effect, due to implementation delays for some species.

1.3. Manta ray regulations in Indonesia

Of the elasmobranchs recently listed on CITES, manta and devil rays (collectively mobulids) have been increasingly targeted in recent decades to meet emerging demand for their pre-branchial appendages, commonly referred to as gill plates, in Chinese medicine markets (Couturier et al., 2012; Hau et al., 2016; O'Malley et al., 2017; Ward-Paige et al., 2013). Retailers market mobulid gill plates for general health maintenance and 'detoxification', and as a cure for various throat and skin ailments, amongst other afflictions. However, these purported medical claims are not supported by medical research or documented in traditional Chinese medicine (TCM) texts (O'Malley et al., 2017).

Manta rays (*Mobula alfredi* and *M. Birostris*) are particularly large, slow-growing, long-lived species, which makes them especially vulnerable to trade-driven extinction (McClenachan et al., 2016). As a nation experiencing intense fishing pressure, and a major source country for the global trade in elasmobranchs, Indonesia is an international priority for elasmobranch conservation (Dent and Clarke, 2014; McClenachan et al., 2012). Over the past decade Indonesia has also been recognised as one of the world's largest mobulid producers (in terms of wild-caught capture production), and a major supplier of gill plates to consumer markets in China, Hong Kong and Singapore (Dent and Clarke, 2014; Hau et al., 2016; O'Malley et al., 2017).

Following growing international concern regarding their overexploitation, both manta ray species were added to Appendix II of CITES in at CoP 16 in 2013. In 2014, the Indonesian Ministry of Marine Affairs and Fisheries (MMAF) issued a Ministerial Decree declaring both species of manta ray fully protected, thus prohibiting all fishing, retention, utilisation and trade of mantas throughout the country's entire exclusive economic zone (MMAF No.4/KEPMEN-KP/2014). This is stronger than the convention requires - CITES stipulates that utilisation and trade is permitted for Appendix II species, provided it is legal and sustainable. However, full prohibition was deemed appropriate by the Government of Indonesia (GoI), due to the value of manta rays for tourism (O'Malley et al., 2013) and the challenge of ensuring sustainable use for such a large-bodied species (Dulvy et al., 2014a,b). Since this decree, GoI, with the support of several Non-Governmental Organisations (NGOs), has made a concerted effort to implement and enforce this regulation, including the arrest and prosecution of more than 20 illegal manta ray traders between 2014 and 2016 (present study). Herein these efforts are collectively referred to as "the manta regulation and associated implementation actions".

Despite these efforts, international demand for manta products persists (Hau et al., 2016; O'Malley et al., 2017) and there are reports of continued fishing and trade in some locations (present study), suggesting the survival of a now illegal industry. As such, this represents an important and interesting case study for evaluating the impact of wildlife use and trade regulation in the face of practical and technical challenges, which are typical of most international wildlife trade assessments and elasmobranch species. Acknowledging these overarching challenges for assessing the impact of wildlife trade regulations, the specific gaps and needs for elasmobranchs, and the pertinent case study of manta rays in Indonesia, we seek to:

1) Highlight key challenges for assessing the impact of wildlife use and trade regulations, which are relevant to most species and contexts;
2) Offer a potential solution, through outlining an integrated framework and multi-stage process, for assessing the impact of wildlife use and trade regulations, which is applicable for any regulation, species or country context;
3) Demonstrate the application of this framework through a worked example, based on the case of manta rays in Indonesia, and available data for this case study. In doing so, we also provide the first attempt to assess the impact of use and trade regulations on manta ray mortality in a source country, and offer recommendations for future formulation, implementation and evaluation of similar regulations for other countries or species.

2. Key challenges in assessing the impact of wildlife regulations

Understanding how regulations on the use and trade of wildlife impact specie's status in the wild is challenging. Oversimplification can lead to flawed conclusions regarding the impact of regulations (Underwood, 2016), which can in turn lead to ill-informed management actions and negative consequences for wildlife. Experimental and quasi-experimental research designs, which use counterfactuals to statistically control for confounding factors, are often seen as the gold standard of impact evaluations (Ferraro, 2009). However, the complex, dynamic context of international wildlife trade limits the feasibility of adopting such research designs (Margoluis et al., 2009). Key challenges include data paucity, system complexity and bias.

2.1. Data paucity

For most wildlife that is utilised and traded there is a lack of integrated species-specific data collection across their spatially and temporally diffuse trade chains, and wide variety of products (Phelps et al., 2010; Smith et al., 2009). Gaps, inconsistencies and lack of granularity in trade data hinder reliable identification of trends in trade and trade-driven mortality of species across space and time (Cawthorn and Mariani, 2017; Phelps et al., 2010), and before-after regulation comparisons.

Where data are available, methodological inconsistencies in data collection (e.g., sampling effort, enforcement effort) or data recording (e.g., units of measurement, taxonomic granularity) can limit reliable quantification of trade volumes, and mask or confound the effects of regulation (Cawthorn and Mariani, 2017; Foster et al., 2016; Gavin et al., 2010). For example, although the CITES trade database is the world's largest available data set on international trade in CITES listed species, and serves as the means for monitoring implementation of the Convention, it relies on annual reports submitted by Parties. However, compliance with CITES reporting requirements is highly varied across nations, in terms of report submission and report quality. There are biases and inconsistencies in detection rates and seizures (based on monitoring and enforcement) and reporting rates (i.e., the information submitted to CITES), which are related to capacity, governance and political will (D'Cruze and Macdonald, 2016; Underwood et al., 2013; UNEP-WCMC, 2013). Illustrating this, only one commercial manta ray import to Hong Kong (from Sri Lanka in 2015) has been recorded in the CITES trade database since manta rays were listed on Appendix II in 2014, and no commercial manta imports have been recorded for China (UNEP-WCMC, 2018). However, a market survey conducted in China and Hong Kong in 2015–2016 found hundreds of stores selling manta ray gills, with sourcing reported from more than 12 countries, including Indonesia (Hau et al., 2016). Many of these countries have no reported commercial manta exports or seizures according to the CITES database, and no reported landings according to FAO production statistics (Dent and Clarke, 2014; UNEP-WCMC, 2018).

2.2. System complexity

Wildlife trade chains are complex, dynamic and heterogeneous ('t Sas-Rolfes et al., 2019; Harfoot et al., 2018). Systems are spatially and temporally diffuse, with interactions between wildlife resources and several layers of wildlife users (e.g., primary exploiters, traders, consumers), and many drivers influencing trade chains at different levels and scales (Burn et al., 2011; Underwood et al., 2013). These can be regulatory drivers (e.g., trade bans, species protection, fisheries management regulations) or exogenous drivers (e.g., economic growth in consumer countries, markets for substitute goods), which may be further confounded by natural variation or inherent randomness in the socio-ecological system (e.g., population stochasticity of the wildlife resource, market variation (Fig. 2)). This means there are several potential explanations for change that need to be disentangled, and outcomes are likely to vary for different scales and contexts. What is more, since impacts can rarely be isolated to treated and untreated groups, comparative groups for experimental or quasi-experimental statistical matching can be difficult to identify and use (Margoluis et al., 2009).

2.3. Bias

Once use and trade of wildlife becomes regulated, it can create legal grey areas, black markets and lead to unintended or perverse impacts on supply and demand (Challender et al., 2015a). The illicit nature of criminalised parts of the trade cause unsystematic and/or hidden biases in monitoring data, which can be context-dependent. Examples include response or social bias in direct questioning, due to fear of retribution; observation bias, whereby the presence of observers discourages illegal activities; and effort or corruption bias by enforcement agencies, which can result in deliberate underreporting and ineffective enforcement (Table 2, and Gavin et al. (2010) for a full review). Since most data collection is not mandatory or reviewed by third parties, data is likely to be inconsistent between countries and species, and may be biased by the interests of the parties collecting the data. These biases are often difficult to control for, and when hidden may mask or confound the effects of regulation and enforcement (Gavin et al., 2010).

2.4. Oversimplification can lead to flawed conclusions

We can illustrate the effects of these challenges through the well-researched example of ivory seizures and elephant mortality. An observed change in reported ivory seizures for a given country could be used as empirical evidence to assess the effectiveness of elephant conservation interventions. A simplistic analysis could assume that a decrease in ivory seizures is due to successful conservation efforts leading to fewer poached and trafficked elephants. Alternatively, when all biases and hypothetical alternative explanations are considered, this observed trend could be attributed to poor law enforcement, leading to lower seizure rates; non-reporting of seizures; reduced demand in consumer countries; an elephant population crash; or some combination of all of these factors. The inferential weight of the empirical data and the plausibility of these hypothetical explanations will depend on a range of contextual factors within a given country and trade chain, such as corruption, governance, political stability and economic and social development (Underwood et al., 2013).

For example, some analyses have claimed that the legal sale of ivory to China in 2008 was the cause of an increase in illegal killing of elephants, based on data from the Monitoring of Illegal Killing of Elephants program (Hsiang and Sekar, 2016). However, critical assessment of their model, logic and assumptions suggests this was a dramatic oversimplification, which failed to take in to account the multitude of plausible drivers of these trends (Underwood, 2016).

This example highlights the need to examine multiple parts of the trade chain, critically assess data and methods, consider multiple plausible explanations operating on different spatial and temporal scales (Underwood et al., 2013), and the practical challenges of doing so. The same can be said for global reported elasmobranch catch. Observed declines in production could be attributed to improved management, or may be the result of underreporting or overfishing. The most plausible explanation for the cause of these declines depends on broader contextual factors, such as changes in management measures, fishing pressure or socioeconomic drivers such as population size (Davidson et al., 2016).

Overall, there is a need for methodological innovations in collating, analysing and interpreting wildlife use and trade data. Methods are needed which can enable robust inference about the impact of regulation of use and trade on conservation

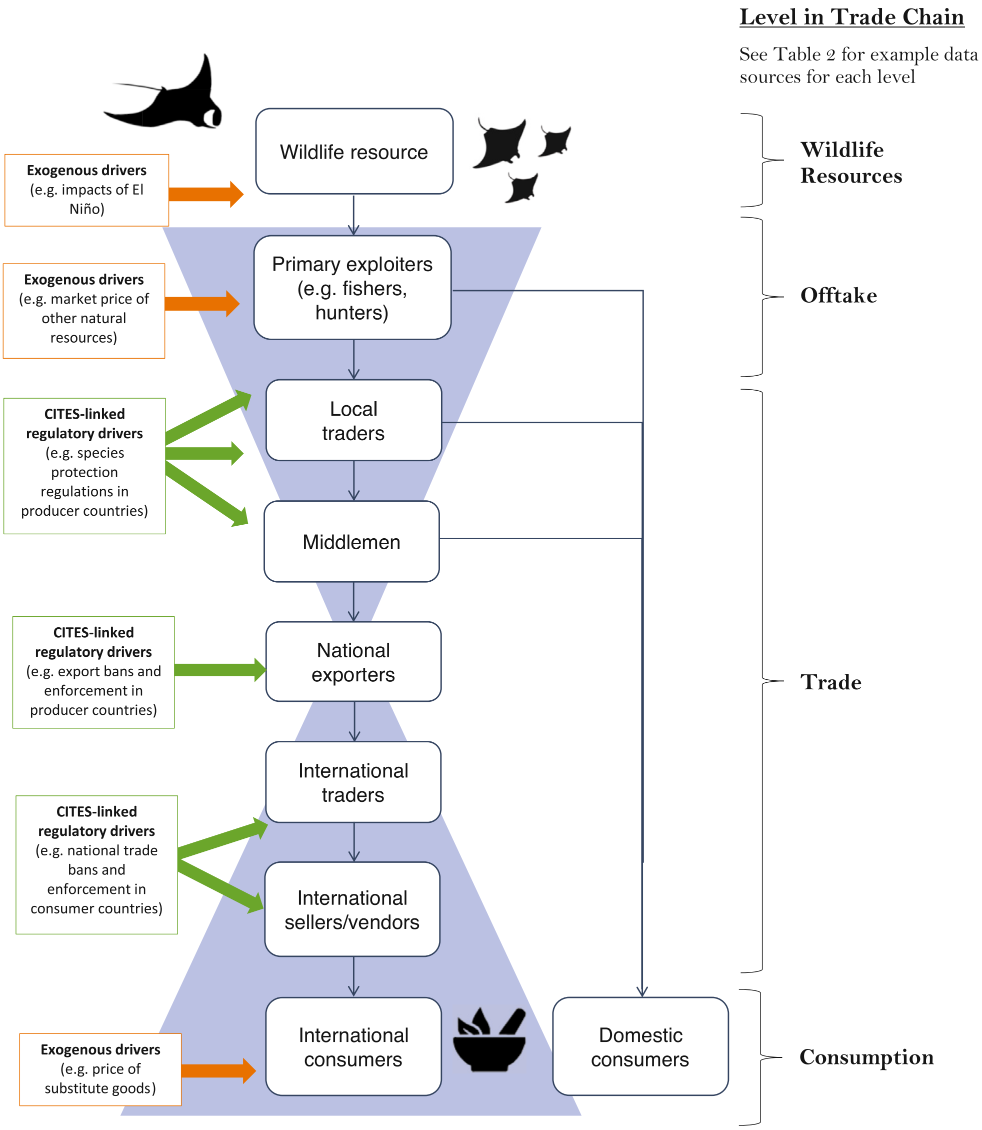

Fig. 2. A simplified conceptual diagram of a wildlife trade chain. The diagram is highly simplified: in reality, there are many actors at each level; and multiple, overlapping routes for the flow of resources, drivers and bi-directional feedbacks. Blue triangles in the background represent the gradual consolidation and dispersal of wildlife products amongst actors at each level e.g., there are many people who harvest and consume wildlife, while products tend to be consolidated by a small number of traders (not to scale). Levels in trade chain listed on the right-hand side correspond to those listed in the flexible data collection framework (Section 3.2).

outcomes, while accommodating and explicitly addressing context, complexity and uncertainty. These methods must also be simple, low-cost and management-relevant if they are to be adopted and useful for governments and practitioners. This is particularly important for elasmobranchs, which are increasingly globally regulated under CITES, yet remain data deficient and exceptionally under-managed worldwide (Dulvy et al., 2017).

3. A potential solution: an integrated framework

Despite these challenges, there are a range of impact evaluation research designs - suiting different budgets, capacities, intervention attributes and data availabilities – from which assessments of wildlife regulations could draw. Different designs have relative strengths and weaknesses with regard to validity, statistical rigour, and the specific research questions they can answer (Margoluis et al., 2009; Woodhouse et al., 2015). Yet for all designs, there is an explicit, systematic process for establishing causal inference and attribution (Ferraro, 2009; Margoluis et al., 2009).

Given the complex characteristics of wildlife use and trade, establishing causal inference about the impact of regulation requires a multi-level systems-based approach, which explicitly incorporates uncertainty. A potential approach is the development and adoption of an integrated framework, which can be used as part of a flexible, multi-stage assessment.

3.1. The benefits of an integrated framework for assessing the impact of wildlife trade regulations

Integrated frameworks are commonly used for assessing conservation issues with multiple, complex causes and effects, and in situations which require incorporation of data from disparate sources (Alcamo et al., 2003; Beauvais et al., 2018; Friedman et al., 2018; Mace et al., 2008; Williams et al., 2008). An integrated framework has multiple benefits, and can be applied to both legal and illegal wildlife use and trade, and cases where legality is different at different levels of the trade chain.

Firstly, an integrated framework can allow for flexible and opportunistic data collection. This is particularly useful in data-poor contexts, as it can help to identify critical data gaps and uncertainties, to in turn guide short- and long-term research priorities.

This flexibility also allows for the combination of qualitative and quantitative methods, to capitalise on the benefits of different research designs and fill data gaps. For example, statistical regressions can be conducted on available quantitative data to identify correlations between regulatory changes and conservation outcomes. Social science techniques can draw on the knowledge of a range of stakeholders, providing high-quality low-cost information, such as local ecological knowledge. This can fill knowledge gaps (Anadón et al., 2009; Drew, 2005; Jones et al., 2008; Leeney and Poncelet, 2015), understand perceived causation from the point of view of those affected (Friedman et al., 2018; Woodhouse et al., 2016), and provide context to or challenge quantitative data by revealing biases. This is advantageous because it can strengthen internal and external validity, by coupling standardised quantitative measures that are comparable to other countries and species with participatory methods to enhance credibility, engagement, inclusiveness and understanding of contextual issues (Woodhouse et al., 2016).

We also emphasise the need for a practical approach, which is ready for use and can be adopted and adapted for different countries, species and contexts. A flexible monitoring framework organised along the lines of a trade chain (Fig. 2) is fit for purpose. It reflects how most existing wildlife use and trade monitoring systems are established, and provides a scaffold on which to draw together different types of data across different sources and scales and build a case for impact assessment. This framework can also be adopted regardless of the causal mechanisms at play by focusing on understanding whether regulatory change *could* have plausibly caused observed trends. It may be supplemented by finer scale case-study evidence that the observed outcomes actually *did* come about because of the regulatory change (Vayda and Walters, 2011), and Theories of Change on *how* and *why* regulatory change made an impact for a given country, place or species. Finally, the trade chain framework also supports triangulation of multiple data sets, to highlight and correct for bias and confounding factors within each data source and trade chain level. For example, quantitative data on exploitation effort (e.g., fishing effort, hunting prevalence) and mortality (e.g., catch) can be used to show statistical trends, while qualitative data can challenge, support or explain observed trends, providing information on context, which is needed for establishing causal inference (see Section 3.2.3).

3.2. Using an integrated framework in a multi-stage assessment process

3.2.1. Defining the question

The assessment process should begin with defining the purpose of the impact evaluation, and therefore the overarching research question. For example, the purpose may be internal learning within a country or institution, to inform adaptive management and future regulatory change, or may be to demonstrate outcomes externally to advocate for continued or similar work (Woodhouse et al., 2016). Depending on the purpose, potential research questions include: "Has regulation X had a positive impact on the conservation status of species Y?"; "What impact has regulation X had on outcome Y?", which may involve quantification of impact; or "How or why has regulation X had an influence on outcome Y?", which involves understanding the mechanisms for change. In the case of assessing the impact of a national-level regulatory change, a realistic research question could simply be "Has regulation X had a positive impact on the conservation status of species Y?" (Table 1). Setting the research question at the beginning of the process is important, as the question will influence subsequent data needs and research design. However, an exploratory or iterative process may also be adopted, with the question revised once data limitations are revealed in stage 2 (Table 1). The research question and its potential implications may also influence the burden of proof, and acceptable levels of uncertainty for drawing a conclusion. For example, if the research is largely descriptive and its potential downside risks are low (e.g. "Has regulation X had a positive impact on the conservation status of species Y in country Z?"), the burden of proof and acceptable levels of uncertainty will be different than a question with large downside risk (e.g. "Should regulation X be implemented for species A and B in countries D and E?"). The precautionary principle should be adopted as best practice when establishing the burden of proof (Cooney and Dickson, 2012).

Table 1
A multi-stage assessment process for understanding the impact of wildlife trade regulations, using an integrated framework for data collection and process tracing techniques for causal analysis.

Stage in the assessment	Key questions/considerations
1. Defining the question	What is the purpose of the impact assessment? What question(s) are we trying to answer? What are the potential implications of the research, and the associated risks? What is the (precautionary) burden of proof and acceptable level of uncertainty, given the potential risks?
2. Collating data	What empirical data is available to answer my question? How was the data collected, and what are the pros, cons and key sources of bias and uncertainty within these data? What are the key data gaps? Can they be filled within the scope of the impact assessment, and how?
3. Establishing causal inference and attribution	
3.1. Analyzing empirical data	What are the *observed* trends in available data?
3.2. Assessing inferential weight	Are these observed trends *real*? Do I have any reason to distrust the data, based on broader contextual considerations?
3.3. Assessing relationships between variables	Is there a *correlation* between regulation X and observed outcome Y?
3.4. Formulating alternative explanations	What are some plausible explanations for the observed trends?
3.5. Assessing alternative explanations	If X and Y are correlated, is this because X caused Y, or because some other variable(s) caused both X and Y? What evidence supports or refutes each explanatory claim?
4. Drawing a conclusion	Based on all observed trends, their respective inferential weight, and alternative plausible explanations, is it reasonable to conclude that outcome Y was *caused* by regulation X? Is regulation X the simplest and most likely explanation? Has the burden of proof defined in stage 1 been sufficiently met by the evidence and associated uncertainties?

3.2.2. Collating data

Once the question has been defined, an integrated framework can be adopted for collating data. Preliminary desk-based research and stakeholder consultation can help to identify available data, as well as improving understanding of key themes, cases, sources and contextual considerations at different levels of the trade chain. The types of data that can be used will depend on the species, country and context in question, though it will be useful to gather as much data as possible, across different spatial and temporal scales, and throughout the trade chain. Preliminary collation of available data can also help to identify data gaps, and how they can be filled. Additional primary data collection may also be conducted at this stage, if required and feasible.

Identifying pre-regulation baseline data for comparison with the post-regulation state is particularly important. A full review of all methods and data available for wildlife regulation impact assessments is outside the scope of this paper, however some examples of the simplest and most widely-available data are summarised in Table 2 (see Gavin et al. (2010) and Solomon et al. (2015) for reviews of methods to monitor illegal use of natural resources). Each method and dataset has unique pros and cons, and sources of context-specific bias and uncertainty (Table 2). These need to be explicitly considered during data gathering and analysis, with a parallel assessment of appropriate metadata (e.g., who collected the data, why, and with which methods/limitations?), and the history of implementation and enforcement, both of which will play a role in shaping biases in data. At this stage it can also be helpful to think about analytical methods to identify trends and correlations; and comparable species, countries or sites that might be suitable for case comparison or natural experiments.

3.2.3. Establishing causal inference and attribution

Once all available data have been gathered, the framework provides a basis for analysing and interpreting datasets, and systematically establishing causal inference using process tracing. Process tracing is based on abductive reasoning, with systematic examination of diagnostic pieces of evidence that are used to judge competing explanatory claims for an observed state or outcome, and inference as to the best (i.e., simplest and most likely) explanation (Bennett, 2010; Vayda and Walters, 2011). There are two overarching steps in process tracing: 1) establishing a correlation between an observed outcome (Y) and a possible explanatory event (X), which in this case are a conservation outcome and regulatory change, respectively. 2) Establishing a plausible causal link between X and Y. That is, whether X and Y are correlated, and if so, is this because X had a causal effect on Y, or because some additional variable(s) or event(s) caused Y (Bennett, 2010).

In the complex reality of wildlife trade chains, an observed state is likely to have been caused by an interplay of variables, such that accurate calculations about causal relationships between X and Y will not be possible. As such, the assessment is not quantitative. Rather, it is analogous to how a doctor diagnoses an illness, using a combination of diagnostic tests (which are analogous to datasets and correlations) and case history (which is analogous to metadata and context). Causal reasoning may be made through considering natural experiments or counterfactual thought experiments (Freedman, 2008; Vayda and Walters, 2011), with development of several hypothesised causal explanations, and assessment of the inferential weight of evidence for each possible explanation (Bennett, 2010; Punton and Welle, 2015; Woodhouse et al., 2016). Assessments can be based on a combination of systematic procedures, involving analysis of contextual factors, such as occurrence of events and outcomes in time and space, and metadata, such as how data were collected and the credibility of data sources; and intuition, based on knowledge and experience (Vayda and Walters, 2011).

Table 2
A flexible framework for collating, analyzing and interpreting wildlife use and trade data from different scopes, scales and sources throughout the trade chain. The example datasets are not exhaustive, but represent those for which data may be most readily available or methods that are feasible to implement.

Level in trade chain	Example data sources	Example analytical methods to identify trends	Example contextual considerations that influence inferential weight	Example alternative (i.e., non-regulatory) hypotheses for observed trends
Wildlife resource	Population estimates, based on censuses or capture-recapture studies	Modelling, statistical regression; quasi-experimental designs using statistically matched species/populations, case comparison	Measurement error, sampling error, model uncertainty	Changes in population status due to natural stochasticity or exogenous environmental factors
Offtake	Prevalence or frequency of harvesting behaviour based on direct questioning or sensitive questioning techniques with primary exploiters.	Ranking and scoring, statistical regression, descriptive analysis	Sample sizes, response and social bias, subjective judgement	Interviewees are giving dishonest answers; changes in effort due to exogenous socio-economic factors (e.g., increased profitability of other species or incomes source) or environmental factors (e.g., variation in population); changes due to offtake becoming clandestine and/or non-reporting.; changes in modus operandi of resource users, such as shifts to un-monitored sites (i.e., displacement)
	Overt direct or indirect observation (e.g., fishers/hunters encountered per unit monitoring effort)	Statistical regression	Human behaviour may be sensitive or clandestine, non-human victims cannot report (i.e., silent victims)	
	Estimated mortality or mortality per unit harvesting effort based on overt direct or indirect observation at key sites (e.g., landings data, poaching rate, carcass ratios)	Modelling, statistical regression, quasi-experimental designs using statistically matched species or populations, case comparison	Behaviour may be sensitive or clandestine, may be possible to conceal mortality	
	Estimated total mortality based on government reported production statistics	Statistical regression, descriptive analysis	Not available or high non-reporting bias if harvesting is illegal, government data rarely species-specific	
Trade (domestic and international)	Prevalence or frequency of trading behaviour based on direct questioning or sensitive questioning techniques with traders	Ranking and scoring, statistical regression, descriptive analysis	Sample sizes, response and social bias, subjective judgement	Changes due to exogenous socio-economic factors (e.g., reduced demand or profitability) or environmental factors (e.g., variation in population); changes due to trade becoming clandestine and/or non-reporting; changes in modus operandi, such as shifts to un-monitored locations (i.e., displacement)
	Prevalence or frequency of trading behaviour covert direct observation using informants/intelligence networks	Statistical regression, descriptive analysis	Source credibility, corruption, knowledge/effort gaps	
	Prevalence or frequency of illegal trade based on arrests or seizures per unit effort from law enforcement data	Statistical regression, modelling	Enforcement effort and credibility, corruption, knowledge/effort gaps	
	Prevalence or frequency of trading behaviour based on direct or indirect observation using market surveys	Statistical regression, descriptive analysis	Behaviour may be sensitive or clandestine	
	Total exports based on government customs data	Statistical regression, descriptive analysis	Will not be available or will have high non-reporting bias if harvesting of species becomes illegal, government data rarely species-specific	
	Reported international trade based on CITES trade database	Statistical regression, descriptive analysis	Baseline/pre-regulation data will not be available, or will have high non-reporting bias if trade becomes illegal	
Consumption (domestic or international)	Prevalence or frequency of consumption based on indirect observation (e.g., market surveys) or direct questioning or sensitive questioning techniques in consumer markets.	Ranking and scoring, statistical regression, descriptive analysis	Sample sizes, non-response and social bias, subjective judgement	Changes due to exogenous socio-economic factors (e.g., price of substitute goods).

Table 3
Definition of categories for assessing and scoring inferential weight (a) and alternative hypotheses (b).

a)

Inferential weight	Interpretation
High	Data source is highly trustworthy, objective and difficult to corrupt. Uncertainty and/or bias are low.
Moderate	Data source is moderately trustworthy and/or vulnerable to some subjectivity. Contextual considerations give cause to question the observed trends. Uncertainty and/or bias are moderate.
Low	Data source is not trustworthy and/or highly subjective. Contextual considerations give cause to strongly question the observed trends. Uncertainty and/or bias are high.

b)

Likelihood of alternative hypothesis	Interpretation
High	Alternative hypothesis is highly plausible and strongly confounds the empirical data. Alternative hypothesis is more likely to explain observed variation than the regulation.
Moderate	Alternative hypothesis is moderately plausible and may confound the empirical data. Alternative hypothesis likely to explain some of the observed variation, though not sufficient to explain all.
Low	Alternative hypothesis is implausible or does not have a strong confounding effect on the empirical data. Alternative hypothesis unlikely to explain the observed variation, or will explain only a small amount.

Process tracing lends itself well to understanding the impact of regulations. It can be conducted post-hoc, without prior research planning. It also employs methods for empirically assessing causal hypotheses when experimental control and replication are not feasible, and there are numerous and complexly interacting causes (Punton and Welle, 2015; Vayda and Walters, 2011; Woodhouse et al., 2016).

When adopting process tracing to establish causal inference for wildlife regulation impact assessment, it is necessary to 1) analyse observed trends in empirical data, the methods for which will vary depending on the type of data available; 2) apply an inferential weight to each source of data, based on an explicit judgement of bias and uncertainty (Table 3); and 3) develop alternative hypotheses for explaining observed trends, and the likelihood of these alternative explanations being true. A simple, low-tech approach for assessing inferential weight and the likelihood of alternative explanations is using an informed judgement with a simple high-to-low or traffic light categorisation system (Table 3), as is commonly used in risk assessments in the biological sciences (e.g., the IUCN Red List Assessment (Mace et al., 2008); the World Organisation for Animal Health risk assessment (Beauvais et al., 2018); ecological risk assessments for fisheries (Cortés et al., 2010)). Several methods also exist for more quantitative assessment of uncertainty, such as the IDEA protocol (Hemming et al., 2018), Value of Information Analysis and Bayesian statistics (Milner-Gulland and Shea, 2017), which could be adapted for use in this process.

3.2.4. Drawing conclusions

Once these steps have been completed for each individual dataset it is necessary to draw together all available evidence to make an assessment. For example: given all observed trends, their respective inferential weights, and alternative plausible explanations, is it reasonable to conclude that outcome Y was caused, at least in part, by regulation X? This is an informed judgement, based on all available information, though could be supported through a quantitative ranking and scoring approach. For example, an assessor could assign scores to each type of evidence in the framework based on its attributes (i.e., observed trend, inferential weight and the likelihood of alternative explanations). A composite 'score', indicative of the overall direction and strength of the available evidence, could then be calculated. At this stage it is also important to check whether the burden of proof (defined in stage 1, as per the question and the risks associated with its implications) has been sufficiently met to draw an appropriate conclusion. This is particularly important if it will inform a subsequent policy or intervention recommendation.

4. A worked example: manta rays in Indonesia

4.1. Case study: Background

In 2014, the Indonesian Ministry of Marine Affairs and Fisheries (MMAF) issued a Ministerial Decree to declare both species of manta ray fully protected, thus prohibiting all fishing, retention, utilisation and trade of mantas throughout the country's entire exclusive economic zone (MMAF No.4/KEPMEN-KP/2014, which is referred to herein as "the manta regulation").

Evaluating the impacts of the manta regulation necessitates an assessment of trends in manta catch, utilisation and trade, and a comprehensive causal analysis of these trends. However, this presents several practical and technical challenges which are typical of most international wildlife trade assessments and elasmobranch species, including: bias, complexity and data paucity. Relevant data are available, but they are methodologically inconsistent, temporally and spatially disparate, and have not been integrated in any meaningful way. This necessitates an exploratory, mixed-methods approach, drawing on multiple datasets at different temporal and spatial scales and levels of the trade chain, with critical assessment of the feasibility and reliability of different methods and sources (Woodhouse et al., 2016, 2015). As such, this case study represents an opportunity to test the integrated framework approach for collating and analysing different types of data, and interpreting the impact of regulation on elasmobranch fishing mortality and onward trade, in a complex, uncertain, data-poor context. The results and lessons learned can also be applied to other taxa and source countries, where similar challenges are faced.

4.2. Case study: Methods

We followed the assessment process and integrated framework as outlined in Section 3.2. The Ethics Committee for the MSc in Conservation Science, Faculty of Natural Sciences, Imperial College London, approved this research.

4.2.1. Defining the question

Given the challenges of quantifying the impact of the manta regulation we focused on understanding whether change has occurred, and whether the manta regulation contributed to this change (i.e., answering the question: “Has the manta regulation and associated implementation actions had a positive impact on the conservation status of manta rays in Indonesia?”.

4.2.2. Collating data

We adopted the integrated framework outlined herein (Table 2) for collating data. The framework was populated through an iterative process of targeted literature review, consultation with non-sensitive stakeholders (defined as those not directly engaged in manta ray trade e.g., NGOs, researchers, government) and an expert-opinion led survey (S1) to identify the most appropriate and readily available empirical data (Table 4).

For fishing effort, catch and local trade, two case study sites were identified with pre-regulation and post-regulation data: Tanjung Luar (Lombok, West Nusa Tenggara) and Lamakera (Solor, East Nusa Tenggara). The two sites are of historic scale and importance as mobulid fisheries (Lewis et al., 2015), have been subject to manta ray law enforcement efforts, and represent different case types in terms of physical and social context, allowing for case comparison (Yin, 2003) (S2). Intelligence and law enforcement data was available for several locations throughout the country, and price data was available for local-, national- and international markets (Table 4, S3).

4.2.3. Establishing causal inference and attribution

4.2.3.1. Analysing empirical data.

4.2.3.1.1. Wildlife resource. Insufficient population modelling and post-regulation data meant that robust statistical analysis of trends in manta ray encounters was not possible in the scope of this assessment.

4.2.3.1.2. Offtake. Catch data was analysed using statistical regression of available landings data from 2013 to 2016. We conducted a multi-step events-based analysis using generalised linear models (GLM) to identify trends, and correlation with the manta regulation and associated implementation (S4). Trends in devil ray (*Mobula tarapacana and M. japonica*) landings were used as a comparative indicator of exogenous factors such as seasonality and market fluctuations, since they are ecologically similar and exploited for the same market, but were not under any domestic or international regulation at the time of the study. We applied a post-hoc generalised linear hypothesis test (GLHT) to statistically compare trends for the two species groups. We also extrapolated estimated total annual landings for both sites using a modified version of the methods in White et al. (2006) to provide an overview of trends in total mortality over time and comparison with historic data (S4). We used RStudio Version 0.99.489 for statistical analysis.

Due to low respondent numbers and non-random sampling, it was not possible to statistically analyse data from community questionnaires. Descriptive statistics and thematic analyses were used to explore trends, results were used for context and triangulation.

4.2.3.1.3. Trade. Data on manta trading from interview-administered questionnaires was triangulated with intelligence and law enforcement data on known and suspected trading activity.

We combined current and historic price data for manta and devil ray gill plates from international and domestic markets, corrected for inflation, and converted all prices to US dollars using exchange rates from Oanda.com. We categorised prices according to site, species and level in the trade chain, and observed trends across levels and sites (S5). We did not conduct statistical analysis, as inconsistent data collection methods will have confounded observed differences.

4.2.3.1.4. Consumption. No new analysis was conducted on consumption, data were taken directly from published studies (Hau et al., 2016; O’Malley et al., 2017).

4.2.3.2. Assessing inferential weight and alternative explanations. Throughout the process we gathered qualitative data on contextual factors pertaining to each case study site, such as fishery and social context, enforcement effort, and attitudes and

Table 4
A summary of readily available empirical data on manta ray exploitation and onward trade in Indonesia.

Method[a]	Dataset	Level of trade chain				Temporal and spatial scale of data	Data sources
		Population	Offtake	Trade	Consumption		
Photo identification	Manta ray encounters	✓				Nusa Penida (Bali), the Gili Islands (West Nusa Tenggara), Komodo National Park and East Flores (East Nusa Tenggara), Raja Ampat (West Papua), Pulau Weh (Aceh), Sangalaki (East Kalimantan). 2006–2016.	Germanov and Marshall., 2014; Manta Watch, 2019; anecdotal information/ unpublished data from various NGOs[c]
Direct questioning using interview-administered questionnaires	Community member questionnaires		✓	✓		Data available in two case study sites: Tanjung Luar and Lamakera, for post-regulation only.	WCS social surveys
Covert direct observation using informers	Intelligence data			✓		Nation-wide[b] intelligence data, post-regulation only.	WCS Wildlife Crimes Unit intelligence database
Overt direct observation using official enumerators	Landings data		✓			Data available in two case study sites: Tanjung Luar and Lamakera, for pre- and post-regulation.	Lewis et al., 2015; White et al., 2006; WCS landings database, Misool Foundation landings database
Indirect observation using market surveys	Price data			✓		Nation-wide[b] and consumer country data available, pre- and post- regulation.	Hau et al., 2016; Lewis et al., 2015; O'Malley et al., 2017; White et al., 2006; WCS market surveys, WCS Wildlife Crimes Unit intelligence database
Law enforcement records from law enforcement actions	Case summaries			✓		Nation-wide[b] law enforcement cases, post-regulation only.	WCS Wildlife Crimes Unit cases database
Indirect observation using market surveys	Sales and stocks of mobulid gills				✓	Data for Singapore, Hong Kong and Guangzhou, 2011–2015	Hau et al., 2016; O'Malley et al., 2017

[a] Categorised based on a modified version of the methods identified in (Gavin et al., 2010).
[b] Noting inevitable gaps/uneven distribution of law enforcement effort throughout the country.
[c] NGO data sources include: the Marine Megafauna Foundation (MMF), Manta Trust/the Indonesia Manta Project, Misool Foundation and Manta Watch.

opinions towards the manta regulation (S2); metadata on each dataset, such as data collection processes (S3) and source credibility; and exogenous influences. This provided commentary on the accuracy, reliability and objectivity of each dataset, which was used to judge respective inferential weights and plausible alternative explanations for observed trends (Bennett, 2010; Woodhouse et al., 2016), as per categories in Table 3.

4.2.4. Drawing conclusions

We populated the integrated framework with available data for each level of the trade chain. This allowed for compilation and triangulation to make an overall judgement of a) whether there had been any real changes correlated with the manta regulation, and b) whether these changes could be plausibly attributed to the manta regulation and associated implementation actions.

4.3. Case study: Results

Photographic identification data of manta rays indicate that healthy populations still exist across Indonesia (Germanov et al., 2019; Germanov and Marshall, 2014; Manta Watch, 2019). However, it was not possible to analyse trends in population status, or correlations with the manta regulation.

Interview and landings data indicate a decline in manta ray fishing effort and catch, with the event-based statistical analysis and interviewee responses indicating a strong correlation with the manta regulation and associated implementation actions. However, differences were noted between Tanjung Luar and Lamakera. The trend and correlation were strong for Tanjung Luar, where observed landings declined to zero in 2015 (highly significant negative difference ($p < 0.001$) between pre- and post-regulation, with a significant interaction between species and regulation), and no fishers self- or peer-reported manta fishing (Table 5, SI). The trend was less strong in Lamakera, where self- and peer-reported behaviour and landings data indicated continued, albeit reduced, manta fishing (Table 5, SI).

Interview, intelligence and law enforcement data indicate a decline in manta trading activity correlated with the regulation. Prior to the manta regulation at least 200 individuals were engaged in elasmobranch trade in Tanjung Luar, although the number specifically engaged in manta ray trading is unclear. Intelligence data indicates two major traders continued

Table 5
A summary of observed trends in all available data throughout Indonesia's manta ray trade chain, along with subjective judgements of inferential weight and the likelihood of plausible alternative explanation.

Level in trade chain	Data set	Observed trade trend	Description	Contextual considerations	Inferential weight	Plausible alternative explanations	Likelihood of alternative explanations
Wildlife resource	Manta encounters from photographic identification and anecdotal information	Unclear	Manta populations are still present across Indonesia, though statistical analysis of trends was not feasible.	Not able to identify trends, information is spatially and temporally patchy.	Low	1. Healthy manta ray populations present in locations that are well protected/were not subject to harvesting pre-regulation	Moderate
Offtake	Community member questionnaires	Decline	Self- and peer-reported manta fishing effort has declined since the regulation. Some interviewees (4) commented they used to catch mantas but have changed because of the regulation.	Sample size is small (n=8); the topic is sensitive non-response and social bias likely to be high; not all responses were consistent.	Low	1. Interviewees are giving dishonest answers 2. Manta fishing effort is declining due to exogenous environmental (e.g. population crash) or socio-economic influences (e.g. reduced profitability as target species) 3. Landings have become clandestine 4. Landings per unit effort declining due to mantra ray population declines/natural stochasticity 5. Observed trends caused by methodological inconsistencies	Moderate
	Landings data	Decline	Estimated total annual catch for mantas has declined in both sites, but has not for devil rays. Statistically significant reduction in manta landing occurrences in Tanjung Luar, but not for devil rays. No significant change detected for Lamakera from May 2015-July 2016, but timeline is short.	Possible to conceal landings so data may not be accurate; data collection methods have not been consistent over time, which may confound trends/makes trend detection challenging.	Moderate		
Trade	Community member questionnaires	Decline	Manta trading has stopped because of the regulation in Tanjung Luar; Manta trading has reduced since the regulation but still continues in Lamakera.	Small sample size (n=5) and the topic is sensitive, but all interviewee answers were consistent.	Moderate	1. Interviewees are giving dishonest answers 2. Manta trading declining due to external influences (e.g. reduced demand in consumer countries) 3. Intelligence information is incomplete and other unknown traders are operating 4. Higher level traders shifted to source from less risky locations	Moderate
	Intelligence data	Decline	No suspected traders operating in Tanjung Luar (3 already arrested), 2 suspected traders yet to be arrested in Lamakera. Other suspected traders still active in some other locations.	Information from credible sources but could be knowledge gaps.	Moderate		
	Law enforcement data	Decline	13 illegal traders arrested nation-wide since 2014. Two from Tanjung Luar, one from Lamakera.	Credible, verifiable source.	High		
	Price data	Decline	Local- and national-level prices have declined since the regulation.	Indirect measure with multiple potential interpretations; inconsistent data collection over time.	Low		
Consumption	Market availability and sales data	Unclear	According to O'Malley et al. 2016, Guangzhou market for gill plates declined sharply between 2011-2015. Conversely, Hong Kong's gill plate sales increased between 2011 and 2015.	Information from peer-reviewed source, but is patchy. Could be knowledge gaps, methodological uncertainty, and non-response or social bias.	Moderate	1. Stocks and sales declining due to in-country conservation campaigns and policies as opposed to source-country conservation efforts (though may also be related to CITES listing) 2. Stocks and sales declining due to changes in price or availability of substitute goods 3. Gill plates sourced from other countries where regulation is weak (i.e. displacement)	Moderate

trading manta products post-regulation, both of whom were arrested and prosecuted in 2015. No further traders had been identified, although secondary and tertiary connections could still exist with other traders operating in West Nusa Tenggera.

In Lamakera, the number of manta product traders prior to the regulation is unknown, but three major manta traders continued trading post-regulation. Of these, two specialised in gill plates, while another specialised in meat. In July 2015 one gill plate trader was arrested and prosecuted, while the two other traders remained operational at the time of study. Mobulid meat is still sold in local markets, though trade routes for gill plates are unclear as enforcement has disrupted known connections between Lamakera and major export hubs. Interviews with fishers and traders in Tanjung Luar and Lamakera were consistent with intelligence information (Table 2). Perceived risk of enforcement for all interviewed traders was that it was "very likely", with comments that they are the "targets" for enforcement and that it is "difficult to cheat" (Table 5, SI).

Local average manta gill plate prices declined in both sites between 2014 and 2015, while prices for devil ray gill plates showed little variation (SI). These local trends contrast national and international trends: national prices rose significantly in 2015, and fell again in 2016 by over 50%; average retail prices in international consumer markets remained relatively stable, with some within-country variability. Indonesia was reported as a major supplier of gill plates to China and Hong Kong by interviewed vendors in 2015 and 2015 (Hau et al., 2016; O'Malley et al., 2017).

A summary of all observed trends is presented in Table 5, alongside commentary on contextual considerations, judgements of each dataset's inferential weight, and plausible alternative explanations for observed trends.

4.4. Case study: assessment

There is strong corroboration across observed trends in different datasets, with all available evidence indicating a decline in manta ray catch and trade, though with varying degrees of strength and reliability. This indicates that the manta regulation and associated implementation actions in Indonesia has contributed to positive conservation outcomes for manta rays. Nonetheless, it remains highly likely that other factors have contributed to the observed trends, such that all observed change in fishing and onward trade cannot be attributed to the manta regulation alone.

For example, in Tanjung Luar, 80% of interviewees ($n = 9$) reported reduced manta fishing and trading because of manta ray protection. These reports are substantiated by an observed step-change in manta ray landing occurrences around the time the regulation was introduced, and by intelligence data suggesting that manta trading at the site has ceased. However, this does not negate the possibility that trade has gone underground. External drivers, such as declining profitability due to reduced consumer demand for mobulids in China (O'Malley et al., 2017), could also be playing a role. However, similar patterns were not observed for devil rays, indicating that a profitable mobulid market still exists.

Given that over 200 manta rays were landed in Lamakera between March 2015 and July 2016, and that two known traders continued to operate, the regulation appears to have been less successfully implemented at this site, although a reduction has been observed. However, the observed differences between Tanjung Luar and Lamakera could be an artefact of insufficient pre-regulation data for Lamakera and more open noncompliance with the regulation. Lamakera also began with a higher baseline catch when the regulation was introduced and enforcement efforts began later, so a time lag in impact is likely. Landings for early 2016 indicate an unprecedented decline in manta ray catch relative to devil ray, which could be the beginning of regulation and enforcement taking effect. However, the data are insufficient to statistically support this: it could also be down to seasonal or climatic fluctuations (e.g., El Niño) or a population decline. It will be useful to update and reassess with additional years of data.

It should also be acknowledged that the trade chain does not operate in isolation in these two sites: effort may have been displaced to other locations that are beyond the scope of this study. For example, anecdotal information from community members in Manggarai, West Manggarai and Ende regencies (East Nusa Tenggara province) indicate that some trade continues in these areas. Data from O'Malley et al. (2017) and Hau et al. (2016) also indicates that manta rays may have been supplied from Indonesia to meet sales and stock figures in Hong Kong and Guangzhou in 2015 and 2016 (Hau et al., 2016; O'Malley et al., 2017). If vendor-reported figures are correct there is a considerable gap between the combined catch figures from Indonesia's two largest mobulid fisheries (~200 in 2015) and the estimated amount of manta ray products sourced from Indonesia for these consumer markets. However, this is difficult to validate, since the non-perishable nature of gill plates limits the feasibility of tracking source populations in space and time (Anderson et al., 2011).

The listing of manta rays on Appendix II of CITES also played a key role as a high-level catalyst for change, which was followed by strong regulatory measures (i.e., full prohibition of utilisation and trade) and implementation (i.e., enforcement). A key factor leading to these conservation outcomes is the commitment of GoI to fully protect manta rays at the national level, and to enforce this law. CITES does not stipulate that Parties fully protect Appendix II species. However, given the value of manta rays for tourism (O'Malley et al., 2013), and the difficulty of ensuring fishing and trade is sustainable for the species (Dulvy et al., 2014a,b), full protection was deemed appropriate. It cannot be expected, however, that all such CITES listings will lead to similar outcomes for other sites, countries and species. For example, despite being listed on CITES Appendix II in 2016, trends in devil ray exploitation have not declined significantly, and in some cases have reportedly increased, suggesting a displacement of effort towards this species, which are not currently protected in Indonesian law. Conversely, it cannot be expected that full species protection and enforcement is the appropriate management response for all taxa, countries and places. For many elasmobranchs listed on Appendix II, particularly species caught incidentally in non-target fisheries and those that play an important role in people's livelihoods and food security, a more nuanced management approach is required. Non-detriment finding studies (NDFs) should provide a scientific-basis for quota-setting, as one component of broader responsible trade and sustainable fisheries management efforts. Regulations and intervention design should be based on species life-history traits, and the fisheries ecology and socio-economic contexts in which they are caught. Context is crucial, and there is no one-size-fits-all management approach (Dulvy et al., 2017).

5. Conclusion

This study highlights challenges and opportunities for evaluating the impact of wildlife regulations in complex, data-poor situations. It presents the first general framework for robust assessment of the impact of wildlife regulation, and the first attempt to understand the impact of regulation stemming from a CITES-listing on manta ray mortality and onward trade in a source country. The process and results also provide lessons learned for future research, impact evaluations, regulatory change and implementation. These lessons are particularly pertinent for elasmobranchs, which urgently require better management, yet such management change is hampered by complexity, data paucity and uncertainty (Dulvy et al., 2017; Lack and Sant, 2011).

5.1. Lessons learned

5.1.1. For research

In the future, assessing the impact of regulations on the utilisation and trade of wildlife would benefit from structured, long-term data collection. Species-specific trade databases, which integrate data across space and time and are ideally in place before a species becomes subject to regulation, could improve consistency and pooling of data. In the past, robust analyses of wildlife trade data have been based on global, longitudinal datasets such as the CITES Elephant Trade Information System (ETIS) and Monitoring the Illegal Killing of Elephants (MIKE) databases. Strengthening low-cost, local-level management-relevant data collection, with an added focus on socio-economic aspects of fisheries and trade, is particularly important for informing elasmobranch management. Collaborative and devolved monitoring, with training and engagement of local stakeholders and managers, could be particularly effective for providing low-cost data, as well as improving participation and social outcomes (Sainsbury et al., 2015; Woodhouse et al., 2016, 2015). Technology and innovative sensitive questioning techniques could help to reduce bias in these data (Langhaug et al., 2010; Nuno et al., 2015). In the absence of improved data collection, modelling can improve robustness and correct for biases and uncertainties (Gavin et al., 2010; Milner-Gulland and Shea, 2017), though may require appropriate metadata (e.g. Underwood et al., 2013).

5.1.2. Consideration for implementation

This study further highlights the need to target multiple levels of wildlife supply chains in a coordinated, strategic, and evidence-based manner, when seeking ot tackle overexploitation of wildlife. This could be supported by mapping and modelling of trade chains, and the complex interactions between different stakeholders and scales. In turn, this could help to ensure resources are invested in impactful interventions in the right places, and limit perverse outcomes. This is particularly relevant in the context of elasmobranch trade, where a large amount of effort is spent on tackling the fin trade, while trade in other commodities and drivers are neglected (Booth et al., 2019b). It is as yet unclear whether closing down the international fin trade would indeed reduce global shark fishing mortality. The complex interactions between different parts of the trade, and the different drivers occurring throughout the trade chain, need to be better understood to inform strategic and cost-effective interventions ('t Sas-Rolfes et al., 2019; Challender et al., 2015a). In the case of manta rays, improving traceability and understanding broader trade characteristics, such as other sources, stockpiles and time lags, are necessary for characterising the trade on a global scale. Continued demand-reduction efforts in consumer countries will also be crucial (Challender et al., 2015a).

The differences between observed outcomes in the two case study sites also highlights the need for locally-appropriate solutions that are nested within broader national and global strategies. The same general solution will not work for all circumstances (Lejano and Ingram, 2010), and Lamakera's engrained cultural traditions of manta hunting (Lewis et al., 2015) likely play a role in the differences in observed outcomes. People in Tanjung Luar do not have a long and engrained cultural attachment to manta ray hunting and local consumption, and have a more diversified fishery, making it easier for people to adapt to regulatory change. Further, Indonesia's manta ray regulation has primarily been implemented by means of enforcement. This gives rise to several potential issues. For example, there are cases where enforcement drives trade underground; undermines positive incentives for conservation; fails to address drivers of conservation problems; and creates social costs, which in turn can undermine legitimacy (Arias, 2015; Arias et al., 2015; Challender et al., 2015a). As our results and previous studies show, the impact of enforcement can be context-dependent. As such, enforcement alone, while perhaps necessary, may not be sufficient to curb manta ray exploitation everywhere in the long-term (Arias, 2015; Arias et al., 2015). Further, since conservation should "do no harm", and enforcement can have disproportionately negative impacts on the poor (Arias, 2015; Arias et al., 2015), there is a moral and practical impetus for considering the socioeconomic dimensions of the manta trade. It will be important to understand what has shaped responses to the regulation in

Tanjung Luar and Lamakera, focusing on the case-specific *how* and *why* of impact. A better understanding of people- and place-specific factors could provide lessons for designing future local-level interventions for CITES implementation which build trust and legitimacy, foster compliance and prevent negative consequences for people and wildlife (Lejano and Ingram, 2010; Oktavia et al., 2018; St John et al., 2010). Social sciences research methods such as scenarios, experimental games and choice experiments could aid intervention design and build trust with communities (Travers et al., 2011; Travers et al., 2016; Travers et al., 2019), while ethnographic studies can help to gain a deep understanding of the different cultural and social contexts in which policies are implemented, and which in turn create and influence conservation outcomes (Infield et al., 2018). Such studies also help to create better intuitions with regard to the causes of change, through building deeper knowledge and experience (Vayda and Walters, 2011).

Finally, while there is always scope for more research, this can lead to trade-offs between 'knowing' vs 'doing' (Knight and Cowling, 2010). In cases where applied research will lead to policy recommendations, and thus have real-world implications for wildlife and people, there are difficult choices around how much information and certainty is enough to act. This is particularly risky in complex and uncertain socio-ecological systems, which may respond to interventions in unexpected ways. Ultimately, the precautionary principle should be adopted, particularly in cases of high uncertainty and high potential downside risk.

5.1.3. The future of wildlife regulation impact assessments

More broadly, this study provides important lessons for wildlife regulation impact assessments in general. Making reasonable inferences about the impacts of wildlife use and trade regulations requires mixed-methods, multiple datasets, and acknowledgement of bias and complexity. These types of evaluations can benefit from: 1) integrated monitoring programmes, which collate spatially and temporally diffuse data, and 2) methodological innovations in the analysis of trade data. Crucially, a combination of qualitative and quantitative methods is required to not only observe trends but to understand the systems and context that create those trends. Such approaches can strengthen causal inference and attribution, and ultimately shape more informed and appropriate interventions into overexploitation of wildlife.

Funding

This work was supported by grants to the Wildlife Conservation Society from the UK Government Darwin Initiative [grant number 110073]; and Vulcan Philanthropy [grant number 110405].

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

The authors are grateful to all individuals and institutions who collected and made available data for use in this analysis. In particular, the Wildlife Conservation Society Indonesia Program provided landings data from Tanjung Luar, law enforcement records, intelligence records, and project records with data from market surveys and community interviews collected by various staff members and consultants. All primary data collection conducted with human subjects by the Wildlife Conservation Society Indonesia Program is reviewed and approved by the WCS Institutional Review Board. Misool Foundation provided landings data from Lamakera for 2015–2016.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gecco.2020.e00953.

References

Alcamo, J., et al., 2003. Ecosystems and Human Well-Being: a Framework for Assessment. Island Press, Washington, DC, USA. www.islandpress.org.

Anadón, J.D., Giménez, A., Ballestar, R., Pérez, I., 2009. Evaluation of local ecological knowledge as a method for collecting extensive data on animal abundance. Conserv. Biol. 23 (3), 617–625. https://doi.org/10.1111/j.1523-1739.2008.01145.x.

Anderson, S.C., Flemming, J.M., Watson, R., Lotze, H.K., 2011. Serial exploitation of global sea cucumber fisheries. Fish Fish. 12 (3), 317–339. https://doi.org/10.1111/j.1467-2979.2010.00397.x.

Arias, A., 2015. Understanding and managing compliance in the nature conservation context. J. Environ. Manag. 153, 134–143. https://doi.org/10.1016/J.JENVMAN.2015.02.013.

Arias, A., Cinner, J.E., Jones, R.E., Pressey, R.L., 2015. Levels and drivers of Fishers' compliance with marine protected areas. Ecol. Soc. 20 (4), art19. https://doi.org/10.5751/ES-07999-200419.

Beauvais, W., Zuther, S., Villeneuve, C., Kock, R., Guitian, J., 2018. Rapidly assessing the risks of infectious diseases to wildlife species. https://doi.org/10.1098/rsos.181043.

Bennett, A., 2010. Process tracing and causal inference. In: Brady, H., Collier, D. (Eds.), *Rethinking Social Inquiry* (Second). Rowman and Littlefield, Maryland. http://philsci-archive.pitt.edu/8872/1/Bennett_Chapter_in_Brady_and_Collier_Second_Edition.pdf.

Berkes, F., 2007. Community-based conservation in a globalized world. Proc. Natl. Acad. Sci. U. S. A. 104 (39), 15188–15193. https://doi.org/10.1073/pnas.0702098104.

Booth, H., Squires, D., Milner-Gulland, E.J., 2019a. The Mitigation Hierarchy for Sharks : A Risk-Based Framework for Reconciling Trade-Offs between Shark Conservation and Fisheries Objectives, pp. 1–21. https://doi.org/10.1111/faf.12429. July.

Booth, H., Squires, D., Milner-Gulland, E.J., 2019b. The neglected complexities of shark fisheries, and priorities for holistic risk-based management. Ocean Coast Manag. 182 (September), 104994. https://doi.org/10.1016/j.ocecoaman.2019.104994.

Broad, S., Mulliken, T., Roe, D., 2002. The nature and extent of legal and illegal trade in wildlife. In: Oldfield, Sara (Ed.), The Trade in Wildlife Regulation for Conservation. Earthscan Ltd, London, pp. 3–22. https://doi.org/10.4324/9781849773935-11.

Burn, R.W., Underwood, F.M., Blanc, J., 2011. Global trends and factors associated with the illegal killing of elephants: a hierarchical bayesian analysis of carcass encounter data. PloS One 6 (9), e24165. https://doi.org/10.1371/journal.pone.0024165.

Cawthorn, D.-M., Mariani, S., 2017. Global trade statistics lack granularity to inform traceability and management of diverse and high-value fishes. Sci. Rep. 7 (1), 12852. https://doi.org/10.1038/s41598-017-12301-x.

Challender, D.W.S., MacMillan, D.C., 2019. Investigating the influence of non-state actors on amendments to the CITES appendices. J. Int. Wildl. Law Pol. 22 (2), 90–114. https://doi.org/10.1080/13880292.2019.1638549.

Challender, D.W.S., Harrop, S.R., MacMillan, D.C., 2015a. Towards informed and multi-faceted wildlife trade interventions. Glob. Ecol. Conserv. 3, 129–148. https://doi.org/10.1016/j.gecco.2014.11.010.

Challender, D.W.S., Harrop, S.R., MacMillan, D.C., 2015b. Understanding markets to conserve trade-threatened species in CITES. Biol. Conserv. 187, 249–259. https://doi.org/10.1016/j.biocon.2015.04.015.

CITES, 2016. List of contracting parties. Retrieved January 7, 2020, from. https://www.cites.org/eng/disc/parties/chronolo.php.

Cochrane, K., 2015. Use and misuse of CITES as a management tool for commercially-exploited aquatic species. Mar. Pol. 59, 16–31. https://doi.org/10.1016/j.marpol.2015.04.015.

Cooney, R., Dickson, B., 2012. Biodiversity and the Precautionary Principle: Risk, Uncertainty and Practice in Conservation and Sustainable Use. Earthscan. https://doi.org/10.4324/9781849770583-12.

Cortés, E., Arocha, F., Beerkircher, L., Carvalho, F., Domingo, A., Heupel, M., Simpfendorfer, C., 2010. Ecological risk assessment of pelagic sharks caught in Atlantic pelagic longline fisheries. Aquat. Living Resour. 23 (1), 25–34. https://doi.org/10.1051/alr/2009044.

Couturier, L.I.E., Marshall, A.D., Jaine, F.R.A., Kashiwagi, T., Pierce, S.J., Townsend, K.A., Richardson, A.J., 2012. Biology, ecology and conservation of the Mobulidae. J. Fish. Biol. 80 (5), 1075–1119. https://doi.org/10.1111/j.1095-8649.2012.03264.x.

Davidson, L.N.K., Krawchuk, M.A., Dulvy, N.K., 2016. Why have global shark and ray landings declined: improved management or overfishing? Fish Fish. 17 (2), 438–458. https://doi.org/10.1111/faf.12119.

Dent, F., Clarke, S.C., 2014. State of the Global Market for Shark Commodities, vol. 590.

Drew, J.A., 2005. Use of traditional ecological knowledge in marine conservation. Conserv. Biol. 19 (4), 1286–1293. https://doi.org/10.1111/j.1523-1739.2005.00158.x.

Dulvy, N.K., Fowler, S.L., Musick, J.A., Cavanagh, R.D., Kyne, P.M., Harrison, L.R., White, W.T., 2014a. Extinction risk and conservation of the world's sharks and rays. ELife 3. https://doi.org/10.7554/eLife.00590.

Dulvy, N.K., Pardo, S.A., Simpfendorfer, C.A., Carlson, J.K., 2014b. Diagnosing the dangerous demography of manta rays using life history theory. Peer J. 2, e400 https://doi.org/10.7717/peerj.400.

Dulvy, N.K., Simpfendorfer, C.A., Davidson, L.N.K., Fordham, S.V., Bräutigam, A., Sant, G., Welch, D.J., 2017. Challenges and priorities in shark and ray conservation. Curr. Biol.: CB 27 (11), R565–R572. https://doi.org/10.1016/j.cub.2017.04.038.

D'Cruze, N., Macdonald, D.W., 2016. A review of global trends in CITES live wildlife confiscations. Nat. Conserv. 15, 47–63. https://doi.org/10.3897/natureconservation.15.10005.

Ferraro, P.J., 2009. Counterfactual thinking and impact evaluation in environmental policy. N. Dir. Eval. 2009 (122), 75–84. https://doi.org/10.1002/ev.297.

Foster, S., Wiswedel, S., Vincent, A., 2016. Opportunities and challenges for analysis of wildlife trade using CITES data - seahorses as a case study. Aquat. Conserv. Mar. Freshw. Ecosyst. 26 (1), 154–172. https://doi.org/10.1002/aqc.2493.

Freedman, D.A., 2008. On types of scientific inquiry: the role of qualitative reasoning. Berkeley, CA, USA. Retrieved from. https://www.stat.berkeley.edu/~census/anomaly.pdf.

Friedman, K., Gabriel, S., Abe, O., Adnan Nuruddin, A., Ali, A., Bidin Raja Hassan, R., Ye, Y., 2018. Examining the impact of CITES listing of sharks and rays in Southeast Asian fisheries. Fish Fish. 19 (4), 662–676. https://doi.org/10.1111/faf.12281.

Gavin, M.C., Solomon, J.N., Blank, S.G., 2010. Measuring and monitoring illegal use of natural resources. Conserv. Biol. 24 (1), 89–100. https://doi.org/10.1111/j.1523-1739.2009.01387.x.

Germanov, E., Marshall, A.D., 2014. Running the gauntlet: regional movement patterns of manta alfredi through a complex of parks and fisheries. PloS One 9 (10), e110071. https://doi.org/10.1371/journal.pone.0110071.

Germanov, E., Bejder, L., Chabanne, D.B., Dharmadi, D., Hendrawan, I.G., Marshall, A.D., Loneragan, N.R., 2019. Contrasting habitat use and population dynamics of reef manta rays within the Nusa Penida Marine Protected Area, Indonesia. Front. Mar. Sci. 6 (APR) https://doi.org/10.3389/fmars.2019.00215.

Harfoot, M., Glaser, S.A.M., Tittensor, D.P., Britten, G.L., McLardy, C., Malsch, K., Burgess, N.D., 2018. Unveiling the patterns and trends in 40 years of global trade in CITES-listed wildlife. Biol. Conserv. 223, 47–57. https://doi.org/10.1016/j.biocon.2018.04.017.

Hau, Y.C.L., Ho, K.Y.K., Shea, K.H.S., 2016. Rapid survey of mobulid gill plate trade and retail patterns in Hong Kong and Guangzhou markets. Hong Kong. Retrieved from. http://www.bloomassociation.org/en/wp-content/uploads/2016/04/Rapid-survey-of-Mobulid-gill-plate-trade-and-retail-patterns-in-Hong-Kong-and-Guangzhou_Revised.pdf.

Hemming, V., Walshe, T.V., Hanea, A.M., Fidler, F., Burgman, M.A., 2018. Eliciting improved quantitative judgements using the IDEA protocol: a case study in natural resource management. PloS One 13 (6), e0198468. https://doi.org/10.1371/journal.pone.0198468.

Hsiang, S., Sekar, N., 2016. Does legalization reduce black market activity? Evidence from a global ivory experiment and elephant poaching data. Cambridge, MA. https://doi.org/10.3386/w22314.

Huxley, 2000. CITES: the vision. In: Hutton, J., Dickson, B. (Eds.), Endangered Species Threatened Convention: the Past, Present and Future of CITES, the Convention on International Trade in Endangered Species of Wild Fauna and Flora, vol. 88. Earthscan Publications Ltd, London, pp. 3–12.

Infield, M., Entwistle, A., Anthem, H., Mugisha, A., Phillips, K., 2018. Reflections on cultural values approaches to conservation: lessons from 20 years of implementation. Oryx 52 (2), 220–230. https://doi.org/10.1017/S0030605317000928.

IPBES, 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. Bonn, Germany. Retrieved from. https://ipbes.net/system/tdf/ipbes_global_assessment_report_summary_for_policymakers.pdf?file=1&type=node&id=35329.

Jones, J.P.G., Andriamarovololona, M.M., Hockley, N., Gibbons, J.M., Milner-Gulland, E.J., 2008. Testing the use of interviews as a tool for monitoring trends in the harvesting of wild species. J. Appl. Ecol. 45 (4), 1205–1212. https://doi.org/10.1111/j.1365-2664.2008.01487.x.

Knight, A.T., Cowling, R.M., 2010. Trading-off 'knowing' versus 'doing' for effective conservation planning. In: Trade-Offs in Conservation. Wiley-Blackwell, Chichester, UK, pp. 273–291. https://doi.org/10.1002/9781444324907.ch15.

Lack, M., Sant, G., 2011. The Future of Sharks: a Review of Action and Inaction. TRAFFIC International and the Pew Environment Group.

Langhaug, L.F., Sherr, L., Cowan, F.M., 2010. How to improve the validity of sexual behaviour reporting: systematic review of questionnaire delivery modes in developing countries. Trop. Med. Int. Health 15 (3), 362–381. https://doi.org/10.1111/j.1365-3156.2009.02464.x.

Leeney, R.H., Poncelet, P., 2015. Using Fishers' ecological knowledge to assess the status and cultural importance of sawfish in Guinea-Bissau. Aquat. Conserv. Mar. Freshw. Ecosyst. 25 (3), 411–430. https://doi.org/10.1002/aqc.2419.

Lejano, R.P., Ingram, H., 2010. Environment: science and policy for sustainable development place-based conservation: lessons from the turtle islands. https://doi.org/10.3200/ENVT.49.9.18-27.

Lewis, S., Setiasih, N., Fahmi, Dharmadi, O'Malley, M., Campbell, S., Sianipar, A., 2015. Assessing Indonesian manta and devil ray populations through historical landings and fishing community interviews. Peer J. https://doi.org/10.7287/peerj.preprints.1334v1. PrePrints.

Mace, G.M., Collar, N.J., Gaston, K.J., Hilton-Taylor, C., Akçakaya, H.R., Leader-Williams, N., Stuart, S.N., 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. Conserv. Biol. 22 (6), 1424–1442. https://doi.org/10.1111/j.1523-1739.2008.01044.x.

Manta Watch, 2019. Manta Watch. Retrieved September 20, 2006, from. https://app.mantawatch.com/.

Margoluis, R., Stem, C., Salafsky, N., Brown, M., 2009. Design alternatives for evaluating the impact of conservation projects. N. Dir. Eval. 2009 (122), 85–96. https://doi.org/10.1002/ev.298.

May, C., 2017. Transnational crime and the developing world. Washington DC. Retrieved from. https://www.gfintegrity.org/wp-content/uploads/2017/03/Transnational_Crime-final.pdf.

McClenachan, L., Cooper, A.B., Carpenter, K.E., Dulvy, N.K., 2012. Extinction risk and bottlenecks in the conservation of charismatic marine species. Conserv. Lett. 5 (1), 73–80. https://doi.org/10.1111/j.1755-263X.2011.00206.x.

McClenachan, L., Cooper, A.B., Dulvy, N.K., 2016. Rethinking trade-driven extinction risk in marine and terrestrial Megafauna. Curr. Biol.: CB 26 (12), 1640–1646. https://doi.org/10.1016/j.cub.2016.05.026.

Milner-Gulland, E.J., Shea, K., 2017. Embracing uncertainty in applied ecology. J. Appl. Ecol. 54 (6), 2063–2068. https://doi.org/10.1111/1365-2664.12887.

Nuno, A., St John, F.A.V., 2015. How to ask sensitive questions in conservation: a review of specialized questioning techniques. Biol. Conserv. 189, 5–15. https://doi.org/10.1016/J.BIOCON.2014.09.047.

Oktavia, P., Salim, W., Perdanahardja, G., 2018. Reinventing papadak/hoholok as a traditional management system of marine resources in Rote Ndao, Indonesia. Ocean Coast Manag. 161, 37–49. https://doi.org/10.1016/J.OCECOAMAN.2018.04.018.

O'Malley, M.P., Lee-Brooks, K., Medd, H.B., 2013. The global economic impact of manta ray watching tourism. PloS One 8 (5). https://doi.org/10.1371/journal.pone.0065051.

O'Malley, M.P., Townsend, K., Hilton, P., Heinrichs, S., Stewart, J., 2017. Characterization of the trade in manta and devil ray gill plates in China and South-east Asia through trader surveys. Aquat. Conserv. Mar. Freshw. Ecosyst. 27 (2), 394–413. https://doi.org/10.1002/aqc.2670.

Phelps, J., Webb, E.L., Bickford, D., Nijman, V., Sodhi, N.S., 2010. Boosting CITES. Science 330 (6012), 1752–1753. https://doi.org/10.1126/science.1195558.

Punton, M., Welle, K., 2015. Straws-in-the-wind, hoops and smoking guns: what can process tracing offer to impact evaluation? Brighton, UK. Retrieved from. http://tinyurl.com/jdk5syt.

Robinson, J.E., Sinovas, P., 2018. Challenges of analyzing the global trade in CITES-listed wildlife. Conserv. Biol. 32 (5), 1203–1206. https://doi.org/10.1111/cobi.13095.

Rowan, M., 2019. CITES – a flawed convention that does wildlife conservation no favours. Retrieved January 7, 2020, from. https://www.dailymaverick.co.za/article/2019-10-23-cites-a-flawed-convention-that-does-wildlife-conservation-no-favours/.

Sainsbury, K., Burgess, N.D., Sabuni, F., Howe, C., Puis, E., Killenga, R., Milner-Gulland, E.J., 2015. Exploring stakeholder perceptions of conservation outcomes from alternative income generating activities in Tanzanian villages adjacent to Eastern Arc Mountain forests. Biol. Conserv. 191, 20–28. https://doi.org/10.1016/j.biocon.2015.06.001.

Shiffman, D., Hammerschlag, N., 2016. Shark conservation and management policy: a review and primer for non-specialists. Anim. Conserv. 19 (5), 401–412. https://doi.org/10.1111/acv.12265.

Simpfendorfer, C., Dulvy, N.K., 2017. Bright spots of sustainable shark fishing. Curr. Biol. 27 (3), R97–R98. https://doi.org/10.1016/j.cub.2016.12.017.

Smith, K.F., Behrens, M., Schloegel, L.M., Marano, N., Burgiel, S., Daszak, P., 2009. Reducing the risks of the wildlife trade. Science 324 (5927), 594–595. https://doi.org/10.1126/science.1174460.

Detecting and understanding non-compliance with conservation rules. *[Special Issue]*. In: Solomon, J.N., Gavin, M.C., Gore, M.L. (Eds.), Biol. Conserv. 189, 1–94. https://doi.org/10.1016/j.biocon.2015.04.028.

St John, F.A.V., Edwards-Jones, G.A., Jones, J.P., 2010. Conservation and human behaviour: lessons from social psychology. Wildl. Res. 37, 658–667. https://doi.org/10.1071/WR10032.

t Sas-Rolfes, M., Challender, D.W.S., Hinsley, A., Veríssimo, D., Milner-Gulland, E.J., 2019. Illegal wildlife trade: scale, processes, and governance. Annu. Rev. Environ. Resour. 44 (1), 201–228. https://doi.org/10.1146/annurev-environ-101718-033253.

Travers, H., Clements, T., Keane, A., Milner-Gulland, E.J., 2011. Incentives for cooperation: the effects of institutional controls on common pool resource extraction in Cambodia. Ecol. Econ. 71, 151–161. https://doi.org/10.1016/J.ECOLECON.2011.08.020.

Travers, H., Clements, T., Milner-Gulland, E.J., 2016. Predicting responses to conservation interventions through scenarios: a Cambodian case study. Biol. Conserv. 204, 403–410. https://doi.org/10.1016/j.biocon.2016.10.040.

Travers, H., Selinske, M., Nuno, A., Serban, A., Mancini, F., Barychka, T., Milner-Gulland, E.J., 2019. A manifesto for predictive conservation. Biol. Conserv. 237, 12–18. https://doi.org/10.1016/J.BIOCON.2019.05.059.

Underwood, F.M., 2016. Critique of Hsiang and Sekar paper linking legal ivory sale to an increase in illegal killing of elephants. Retrieved July 1, 2016, from. http://www.fmunderwood.com/2016/06/20/critique-of-hsiang-and-sekar-paper-linking-legal-ivory-sale-to-an-increase-in-illegal-killing-of-elephants/.

Underwood, F.M., Burn, R.W., Milliken, T., 2013. Dissecting the illegal ivory trade: an analysis of ivory seizures data. PloS One 8 (10), e76539. https://doi.org/10.1371/journal.pone.0076539.

UNEP-WCMC, 2013. A guide to using the CITES trade database - version 8. Cambridge. Retrieved from. www.cites.org.

UNEP-WCMC, 2018. CITES trade database. Retrieved January 20, 2020, from. https://trade.cites.org/.

Vayda, A.P., Walters, B.B., 2011. Introductory essay: pragmatic methods and causal-history explanations. Causal explanation for social scientists. AltaMira Press. Retrieved from. https://www.mta.ca/uploadedFiles/Community/Bios/Brad_Walters/INTRO.to.causal.explanation.reader.pdf.

Vincent, A., Mitcheson, Y. S. de, Fowler, S., Lieberman, S., 2014. The role of CITES in the conservation of marine fishes subject to international trade. Fish Fish. 15 (4).

Ward-Paige, C.A., 2017. A global overview of shark sanctuary regulations and their impact on shark fisheries. Mar. Pol. 82, 87–97. https://doi.org/10.1016/J.MARPOL.2017.05.004.

Ward-Paige, C.A., Davis, B., Worm, B., 2013. Global population trends and human use patterns of Manta and Mobula rays. PloS One 8 (9). https://doi.org/10.1371/journal.pone.0074835.

White, W.T., Giles, J., Dharmadi, Potter, I.C., 2006. Data on the bycatch fishery and reproductive biology of mobulid rays (Myliobatiformes) in Indonesia. Fish. Res. 82 (1–3), 65–73. https://doi.org/10.1016/J.FISHRES.2006.08.008.

Williams, S.E., Shoo, L.P., Isaac, J.L., Hoffmann, A.A., Langham, G., 2008. Towards an integrated framework for assessing the vulnerability of species to climate change. PLoS Biol. 6 (12), e325. https://doi.org/10.1371/journal.pbio.0060325.

Woodhouse, E., Homewood, K.M., Beauchamp, E., Clements, T., McCabe, J.T., Wilkie, D., Milner-Gulland, E.J., 2015. Guiding principles for evaluating the impacts of conservation interventions on human well-being. Phil. Trans. Biol. Sci. 370 (1681), 20150103. https://doi.org/10.1098/rstb.2015.0103.

Woodhouse, E., de Lange, E., Milner-Gulland, E.J., 2016. Evaluating the impacts of conservation interventions on human well-being: guidance for practitioners. London. https://doi.org/10.1017/s0030605316001423.

Worm, B., Davis, B., Kettemer, L., Ward-Paige, C.A., Chapman, D., Heithaus, M.R., Gruber, S.H., 2013. Global catches, exploitation rates, and rebuilding options for sharks. Mar. Pol. 40, 194–204. https://doi.org/10.1016/j.marpol.2012.12.034.

Yin, R.K., 2003. Case Study Research: Design and Methods. Sage Publication, Thousand Oaks, CA.