Scientific Shortcomings in Environmental Impact Statements Internationally

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Abstract

Governments around the world rely on environmental impact assessment (EIA) to provide rigorous analyses and an accurate appraisal of the risks and benefits of development. But how rigorous are the analyses conducted in EIAs, and how do they compare across nations? We evaluate the output from EIAs for jurisdictions in seven countries, focusing on scope (temporal and spatial), mitigation actions, and impact significance determination, which is integral for decision-making. We find that in all jurisdictions, the number of identified significant adverse impacts was consistently small (or nonexistent), regardless of context. Likely contributing to this uniformity, we find that the scopes of analyses are consistently narrower than warranted ecologically and toxicologically, many proposed

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mitigation measures are assumed to be effective with little to no justification, and that the professional judgement of developer-paid consultants is overwhelmingly the determinant of impact significance, with no transparent account of the reasoning processes involved. EIA can be salvaged as a rigorous, credible decision-aiding tool if rigor is enforced in assessment methodologies, regulators are empowered to enforce rigor, and pro-development conflict of interest is avoided.

Key words: environmental impact assessment; review; impact significance; mitigation; environmental impact statement

1. Introduction

Large–scale development is a hallmark of the modern world, providing society with things humans value, but at an environmental cost (Crutzen 2006; Hellweg and Milà i Canals 2014). To navigate this trade-off, many governments rely on the process of environmental impact assessment (EIA) to inform development and environmental decision-making by providing an accurate accounting of a development's impacts (Wood 2003). EIA was initiated by the US *National Environmental Policy Act* (NEPA) in 1970, and while the intentions and core elements of EIA are widely shared, this process has been adapted to unique contexts and circumstances around the world(Wood 2003; Jay et al. 2007; NEPA 2007; Glasson et al. 2013). Proponents of EIA refer to it as a "robust," "science-based" approach—terms which carry connotations of credibility and objectivity (Killingsworth and Palmer 2012). But to what degree do EIA practices reflect rigorous research, evidence and analysis as appropriate to the standards in the fields from which they draw?

To answer this question, we examined one of the main outputs of the EIA process—written reports commonly referred to as environmental impact statements (EISs). While the EIA process involves decisions beyond the scope of scientific practice itself, the EIS represents the application of research and evidence in assessing impacts (Jay et al. 2007; Glasson et al. 2013). We examined EISs from regulatory

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jurisdictions in seven locations around the world: British Columbia (Canada), California (United States), Veracruz (Mexico), Brazil, England and Wales, Queensland (Australia), and New Zealand. Our multi-national research focus is uncommon in its combined geographic and conceptual scope, and can provide insight into the state of EIA scientific practice broadly for jurisdictions that engage in similar processes.

In every jurisdiction we sampled there was a general emphasis on EIA contributing to environmental protection and sustainability through mitigation, and for EISs to stand as a transparent public record of assessment (Wood 2003; Glasson et al. 2013). Each EIS in our sample was written by a multidisciplinary team who typically (1) consulted relevant stakeholders (2) established the spatiotemporal scope for the study, (3) determined the potential impacts of the project to valued environmental components (including impacts that might occur in concert with other past, present, and future projects, called cumulative effects), (4) proposed mitigation to avoid, reduce, remedy and compensate identified impacts, and determined the residual impacts that would likely persist after mitigations are applied, and finally, (5) based on all the previous work, determined the importance – or significance – of these residual impacts (Wood 2003). Significance determination is arguably the “bottom line” of all EIS, supplying decision-makers with a final account of the impacts to be weighed against development benefits.

As works of research published for use in decision-making by authorities and the public, we expect EISs to abide by standards of evidence and analysis within relevant disciplines and to be transparent regarding methods and findings. Research disciplines can contribute to EIS methods and analysis in various ways. For example: (1) findings of species ranges and habitat needs from wildlife biology can inform the establishment of spatiotemporal scopes of analysis of impacts on affected species (Long and Nelson 2012); (2) research from environmental toxicology can determine the magnitude and duration of lag effects from decommissioned mines and other developments (Demchak et al. 2004); (3) research into prescriptive methods for public deliberation and decision-making is highly relevant for consultation methods to reflect and respond to stakeholder concerns (Pidgeon et al. 2005; Fishkin 2009); and (4) evidence from restoration ecology can be used to assess the effectiveness and uncertainty of mitigation measures on environmental impacts (Quigley and Harper 2006). Ultimately, the information from these scientific disciplines can be used as important inputs to determine the scope, effects, and uncertainty behind impacts, which can inform the determination of significance of impacts, particularly when significance is partly an identification of irreversible changes to the environment.

In this paper, we evaluate how the current practice of EISs reflects the current state of relevant research fields. We document how often significant impacts are found, and assess the methodological steps that contribute to significance determination, focusing on methodological rigor and transparent communication of methods and results (namely scoping, mitigation assessment, consultation and significance determination methodology). Specifically, we address the following questions: (1) How consistently are potential impacts found to be significant across jurisdictions? (2) Does the scope of an EIS reflect the current state of research practice most relevant to claims made? (3) How robust are the proposed mitigation measures from the point of view of methods and analyses in commensurate field(s)? (4) How is significance determined?

2. Material and Methods

We compiled a database of recent EISs from seven different jurisdictions of the world from diverse continents (excluding Asia and Africa), including British Columbia (Canada), California (USA), Veracruz (Mexico), Brazil, England and Wales, Queensland (Australia), and New Zealand. While many empirical studies of environmental assessments consider a single jurisdiction and specific issue within EISs, we chose to evaluate EISs across multiple diverse jurisdictions looking at the main components of EISs in order to comprehensively assess systematic issues in EISs. In addition, we chose locations for their status as jurisdictions with well-established EIA legislation, the availability of their EISs (EISs are not always publicly available), the language proficiency of our group, as well as geographic diversity in order to explore EIAs broadly. We focused on the EISs alone and not the entire EIA process as the latter involves decisions beyond the scope of scientific practice itself, whereas EISs represent the application of research and evidence in assessing impacts (Jay et al. 2007; Glasson et al. 2013). We reviewed only recent EISs in order to emphasize current legislation, policy, and process in all jurisdictions we investigate (68 in total). The composition of types of projects varies among the jurisdictions in our sample (Table S1), and our analysis allows us to assess scientific quality of reports across broad project types and jurisdictions(Burris and Canter 1997; Hildebrandt and Sandham 2014).]

Jurisdictions we selected include a mix of governance levels (states/provinces and countries) because we chose the most local level at which decisions are made about large-scale industrial projects. Though the EIA process in Mexico is nation-wide (there are no state-level EIA processes), we focused on Veracruz to pair with Brazil as Atlantic coast jurisdictions against British Columbia and California as Pacific coast jurisdictions. We selected the ten most recent EISs from each jurisdiction to focus on current, consistent regulation, policy, and processes. Most EISs were initiated between 2012 and 2015 (one in British Columbia was from 2010 and one from New Zealand was from 2011). The paucity of available EISs in New Zealand led us to review only seven EISs from there, and the high number of EISs in Queensland led us to review 11 EISs. A breakdown of types of projects in each jurisdiction can be found in Table S1.

While we are not exhaustive with the number of jurisdictions that fulfill our criteria of publicly available EISs and well-established EIA regulations, our results are multi-national and have a wide geographic scope (representing four continents). Language restrictions prevented us from evaluating EISs from some parts of the world (such as Asia). Similarly, the time commitment needed to evaluate EISs (documents that are often hundreds to tens of thousands of pages in length) was not feasible for a comprehensive assessment of EISs of every country in the world.

We looked at official guidance documents for each jurisdiction on how to prepare an EIS to ensure that the EISs were conducted according to similar protocol (from predicting impacts, proposing mitigations, and evaluating significance of impacts, Table S5). To ensure that this was the case, we used document analysis (Krippendorff 2004; Bowen 2009) to systematically review the information in EISs and place into specific and predefined categories, a common tool in the review of environmental assessment documents (Lees et al. 2016; Noble et al. 2017). For each EIS, we counted the number of impacts identified in each EIS to estimate the proportion of impacts that were deemed "significant"; distinguishing between recorded project-specific *potential* impacts*, residual* impacts, *cumulative* impacts, and *significant* impacts by relying on the EIS to accurately differentiate these (that is, we took the reports at their word and did not interpret types of impacts for them). We also classified the methods by which significance was determined in broad categories (technical, collaborative, reasoned) as defined by Lawrence (2007). Because of the highly skewed nature of the data on impact frequencies, we used bootstrap 95% confidence intervals of the median (using the bias corrected and accelerated method as it performs reasonably well with low sample sizes(Obuchowski and Lieber 1998; Chernick 2008)) to determine significant differences between jurisdictions. We calculated a global median from all jurisdictions included in our analysis. Where bootstrapped confidence intervals cross the global median, this indicates that there is no significant difference between the jurisdiction and the global median. Analysis was conducted using the boot package in R (Canty and Ripley 2015).

To determine the spatial dimensions for each EIS, we determined largest area investigated by the EISs to assess cumulative impacts (the largest area assessed for all valued components). Where only maps were provided (and data not provided in-text), we calculated area measures from the maps using PlotDigitizer (Huwaldt 2014) and ImageJ (Abràmoff et al. 2004). To assess the suitability of these spatial areas we compared these areas against the published ranges of species that EISs in each jurisdiction consider. We haphazardly sampled a list of animals assessed by our sampled EISs in each jurisdiction (we chose six species assessed in multiple EISs per jurisdiction) and used publicly available resources to acquire data on species ranges (Table S2). We matched the scale of species ranges to the scale at which EISs claim to assess them. We also limited the scale to the boundaries of the jurisdiction. For example, if EISs claimed to assess impacts to specific populations, subspecies, or species, we looked up range data at that scale within the jurisdiction. We made no attempt to interpret EIS author intentions (i.e. if they meant they were assessing impacts to specific populations but only referred to species). We ensured that wildlife was described consistently regarding ecological scales (e.g. populations, species) when wildlife was introduced in the EIS and when impacts to the wildlife were described. Where possible, we used government online resources from each jurisdiction which often described range inside jurisdiction boundaries, or online resources that the government sites provided. Where this was not possible, we used IUCN online resources, and restricted the analysis to the jurisdiction of interest. We recorded if the data was of “area of occupancy” – the area occupied by a taxon – or of “extent of occurrence” – the shortest continuous boundary that encompasses all the known or predicted sites of a taxon’s occurrence (Hurlbert and Jetz 2007). Where available, we recorded the area of occupancy, as this measure is smaller.

To assess temporal scale, we recorded the number of years estimated for construction of the project, the number of years the project was projected to be operational, and the total number of years for which the EIS assessed impacts. The difference between the number of years for impact evaluation and the number of years for operation and construction constituted the number of years past project decommissioning that impacts from the project in each study was considered to contribute to environmental impact. As we noted that mining EISs had the longest post-closure time periods, we focused our analysis on this subset of EISs (N=11). We then collected peer reviewed published data on the number of years post mine closure the effects of acid mine drainage (AMD) have been recorded (Table S3). We contrasted this data with the temporal scope of mining EISs.

To assess the interaction categories of cumulative impacts, we analyzed the EISs' methodology sections and noted how cumulative impacts were described and assessed. If there was any mention of interaction type (e.g. additive, synergistic, antagonistic) we recorded that EIS as having considered that specific interaction type. We also recorded whether the EIS did not specify types of cumulative impacts (but still described their methodology) and whether the EIS did not describe their methodology at all.

To look at the importance of mitigations in significance determination, we analyzed EISs that consider significance before and after mitigations. When an impact considered significant prior to application of mitigations was still considered significant post-mitigation, we noted whether this was because no mitigation was applied to the specific significant impact (e.g. some significant impacts on visual amenity in England and Wales had no mitigations proposed), or because the mitigation was not anticipated to be fully effective. We used this information to compute the ratio of how often mitigation measures changed the significance determination for impacts compared to how often mitigation measures did not. Additionally, we counted the total number of mitigation measures indicated in each report. For each mitigation measure, we assessed whether the language associated with the mitigation was sufficiently vague as to render the mitigation action ambiguous, and recorded the number of mitigations with vague language around implementation or execution. Examples of vague mitigation language include "to the extent possible", "where feasible", "if practical", "will attempt", "explore the possibility of", and "plan to create a plan to mitigate". We also made note of whether the EIS provided evidence for mitigation effectiveness, assessed the effectiveness of proposed mitigations, or acknowledged uncertainty in the proposed mitigations.

We collected data from each EIS on stakeholder consultation. We reviewed each EIS and recorded the level of public engagement that was undertaken according to the typology of participation developed by Hughes (1998) (Table S4). We recorded the most inclusive form of consultation undertaken on behalf of the project. We also recorded the types of stakeholders and affected parties involved in consultation, according to the categories from Hughes (1998).

Multiple members of the author list (9 authors in total) participated in collecting data. To ensure that we minimized among-collector variation, we took measures to standardize data collection. First, all data collectors took part in a short workshop to communally collect data from the initial trial EIS. Second, one of the data collectors who is fluent in all relevant languages (English, Portuguese, and Spanish) either directly collected data, or supervised the collection of data, for all regions, and performed quality control on the completed database. To promote greater consistency, weekly meetings were held for data collectors to ask clarifying questions about coding and compare coding results with each other. A second workshop was also conducted after data collection had begun to help collectors calibrate their approaches with one another. All coders were found to code data in similar ways (e.g. inter-assessor standard deviation in number of significant impacts was 0.33 and inter-assessor standard deviation for mitigations with equivocal language was 0.7). Finally, the group member responsible for the database (a different member than the data supervisor) re-checked the data, paying attention to any data points that seemed to stand out. If any data did stand out, the database manager re-collected the data with the original data collector.

3. Results and Discussion

3.1 Different Places, Same Bottom Line

If impact significance was consistently determined without bias across jurisdictions, we might expect that jurisdictions with similar types of projects and environmental settings would have relatively similar proportions of potential impacts considered significant (Table S1). Absent of a strong pressure leading to low numbers of significant impacts, the high variation in the sample (across geography, diverse suites of development types, and impact numbers) should translate into high variation in numbers of significant impacts within and across jurisdictions. Indeed, the number of potential, cumulative, residual, and residual cumulative impacts reported in EISs varied considerably across jurisdictions. However, regardless of jurisdiction, a consistently small number of potential impacts were considered significant (all bootstrap 95% Cis of the median overlap the global medians of two significant project-specific impacts and zero significant cumulative impacts, Figure 1).

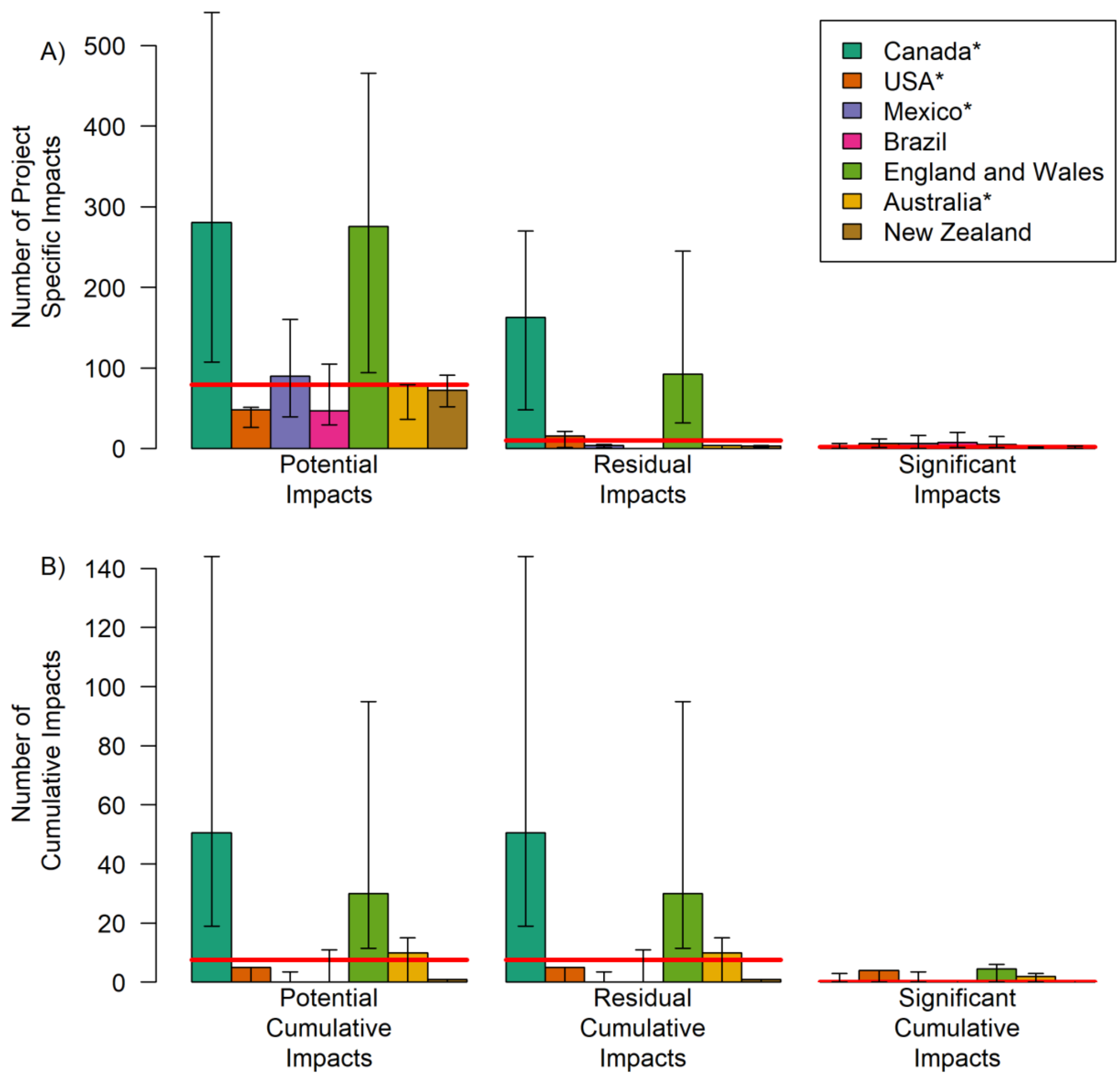

Figure 1. The number of potential impacts, residual impacts, and significant impacts reported in EISs for (A) project-specific impacts and (B) cumulative impacts. Bars represent bootstrap 95% confidence

interval of the medians, and the red lines represent the global medians. EISs were selected from a single state or province within the countries marked with an asterisk (*) in the legend.

One possible explanation for the few significant impacts found across jurisdictions is that the EIA process leading up to preparation of the EIS is a systematic barrier to projects that will likely contribute to significant impacts, allowing only relatively benign projects to undergo significance determination (Wood 2003). An alternative explanation is that the research practices communicated in EISs contribute to bias against finding significant adverse impacts. Below, we discuss the research practices communicated in EISs and whether this alternative explanation is supported.

3.2 Narrowly Addressed Environmental Impacts

Many impact assessments fail to scope their projects in a manner many experts would consider ecologically sufficient or transparent. Fully exploring cumulative impacts to wildlife requires an assessment scope that fully encloses the range of the wildlife at the ecological scale (e.g. populations, species) of interest. For example, a spatial boundary for cumulative impacts assessment might be based on the combined ranges of wildlife populations of value. Where practical or jurisdictional concerns (such as transboundary migration in some species without coordinated management between jurisdictions) might prevent the effective consideration of the full range of wildlife, the spatial scope reported in EISs may be restricted to account for the wildlife within a particular jurisdiction. To illustrate the inadequacy of the spatial scope of the sampled EISs, we collected data on a subset of wildlife assessed in multiple EISs in each jurisdiction, at the lowest ecological scale of interest identified as a valued environmental component. For example, if the valued environmental component was identified as an animal at the species (or specific population) scale, we collected data on their range for the species (or specific population) within the jurisdiction of interest. Though our sample does not allow us to conclude that EIS spatial scope is inadequate for all wildlife assessed, we did find that 98% of the 48 EISs assessing impacts across the wildlife we selected (across all jurisdictions) had at least one wildlife species (or population) that was inadequately spatially scoped for cumulative impacts. In fact, we found that spatial scopes of EISs were considerably smaller than the ranges of species (or specific populations) purportedly assessed in almost all of the sampled EISs (Figure 2A). Only a minority of EISs considered spatial scales comparable to (or greater than) the ranges of species or population units assessed (Figure 2A). In most EISs, the lowest ecological scale explicitly mentioned was the species scale (Table S2). However, if EIS authors were actually assessing impacts to specific populations within these species, they did so without transparently indicating what populations they were evaluating impacts to, or the range size of populations under evaluation, effectively leaving the reader guessing as to the scope of the study. We did note that multiple EISs in every jurisdiction assessed impacts to species given pre-defined spatial boundaries (not determined by wildlife ranges), indicating that the full range of wildlife may not have been considered.

A similar scoping problem was evident in EISs with regard to temporal scales consistent with environmental toxicology. Some projects can affect the environment long past decommissioning, causing lag impacts (Collins et al. 2010). In practice, we found EISs routinely restrict the scope of assessment to well before impacts are likely to cease, as revealed by the illustrative case of mining EISs. Mining EISs in our sample assessed impacts further past decommissioning than other EISs, but even these temporal scopes were generally far shorter than published durations of environmental impacts from acid mine drainage (AMD) after mine closures. Whereas most mining EISs limited their assessment to a period of between zero and four years after mine closure (Figure 2B), independent environmental toxicology studies emphasized that AMD can last decades to centuries past mine closure, even accounting for modern remediation techniques (Demchak et al. 2000; Demchak et al. 2004; Moncur et al. 2006). Out of 26 mining EISs sampled from Queensland, Brazil, and British Columbia, only one (written for a British Columbian mine) had an appropriate temporal scale for assessing AMD impacts past decommissioning. Narrow temporal scoping has further repercussions for future EIA processes, as limiting the number of residual impacts found in one EIS precludes these impacts from becoming relevant inputs to cumulative impact assessments in subsequent EISs.

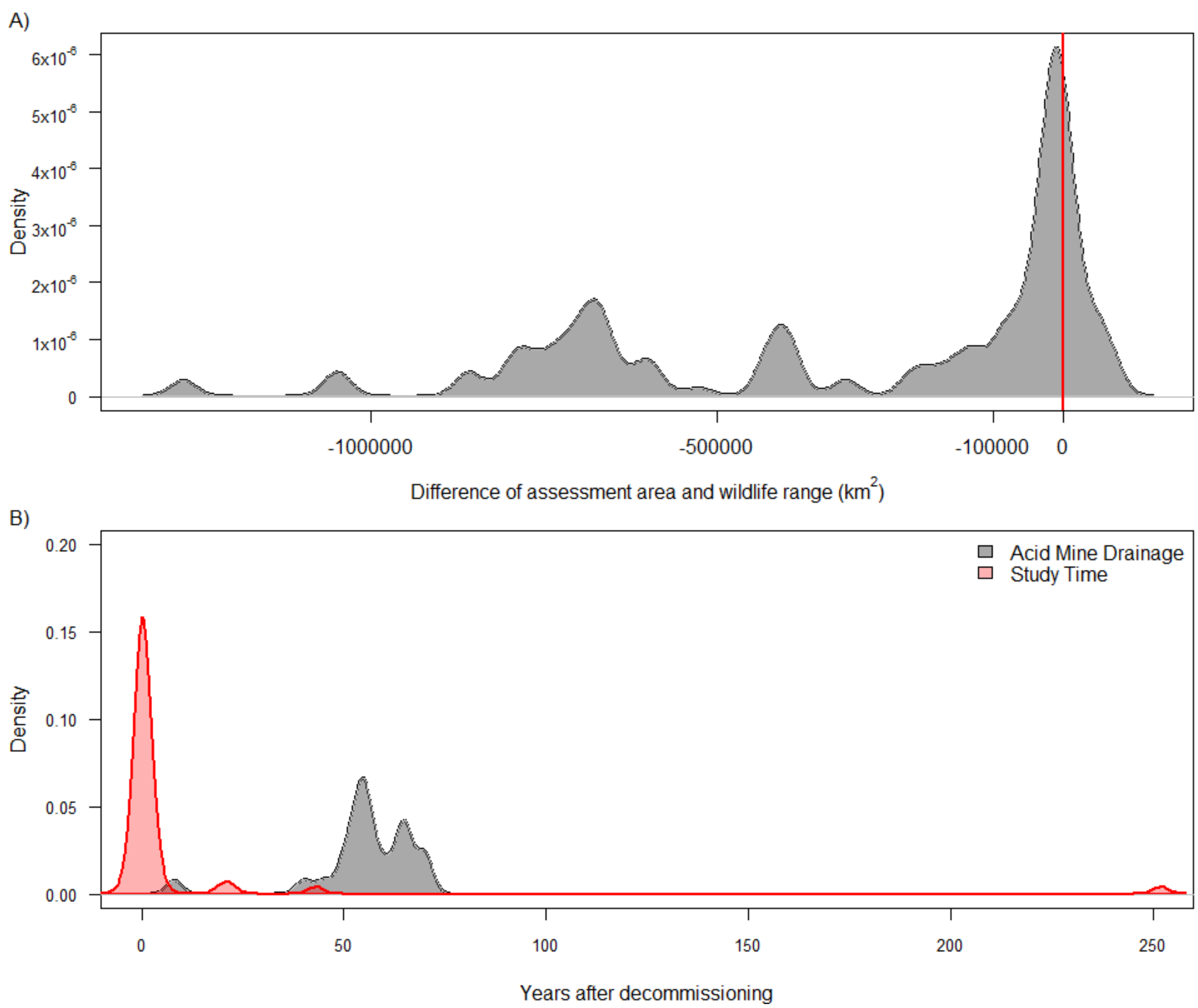

Figure 2. Density histograms showing (A) the difference between the assessment area and wildlife range (in km^2), where negative numbers indicate that wildlife ranges are larger than assessment areas (n = 62 wildlife to EIS comparisons). The vertical red line indicates where assessment area equals wildlife range and (B) the time after mine decommissioning that acid mine drainage impacts ecosystems (in grey) compared with the temporal scope of EISs after mine decommissioning (in red, n = 11 EISs).

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The assessment of potential interactions among impacts was similarly limited and generally opaque. Research suggests that cumulative impacts are often synergistic (where total impact is greater than the sum of individual impacts) or antagonistic (where total impact is less than the sum of individual impact), yet possible synergistic or antagonistic impacts were only explicitly considered in 4% of EISs (Crain et al. 2008; Darling and Côté 2008). Slightly more EISs (15%) had explicitly considered additive impacts (where total impact is equal to the sum on individual impacts), though synergistic or antagonistic impacts were not considered in these EISs. A majority (53%) of sampled EISs were methodologically unclear (methods were provided for assessing cumulative impacts but there was no mention of impact interaction) and 28% provided no methodical explanation for how cumulative impacts were assessed (including every EIS investigated from New Zealand) despite reporting assessment results for cumulative impacts. While we recognize that determining non-additive impact interactions is difficult to accomplish with certainty, the possible existence of these impacts was ignored in 96% of EISs. Additionally, the high percentage (81%) of EISs with unclear or unavailable methods highlights a lack of transparency in assessment. Where methods were available (in 72% of EISs), EIS authors tended to define cumulative impacts as a function of overlapping projects within assessed areas. They did not define cumulative impacts as promoted in the peer reviewed literature, that is, as a function of interacting mechanistic processes linked to specific stressors investigated (Murray et al. 2016). Only 3% of EISs explored cumulative impacts in this manner: for example, explicitly documenting tanker traffic and the effects of underwater noise associated with nearby energy projects. Various frameworks exist to analyze mechanistic processes contributing to cumulative impacts, and these frameworks can be applied even when identifying interactions of impacts is difficult (Knights et al. 2013; Singh et al. 2017).

Ultimately, the limited scope of EISs in space, time, and interactions across impacts all contribute to an avoidably narrow assessment of impacts (Lenzen et al. 2003).

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3.3 Overconfidence in Mitigation

Several of our results demonstrate EISs authors placing high confidence in the effectiveness of mitigation measures – a confidence likely undeserved. In 19% of the EISs we sampled, significance was determined both before *and* after application of proposed mitigations, providing insights into the assumed efficacy of mitigation. These EISs were all from England and Wales, Brazil, Queensland and California. The resulting change in characterization of significance provides some indication of the EIS authors' confidence in the proposed mitigating measures. Out of 505 impacts deemed significant prior to mitigation across these EISs, 80 were ultimately characterized as significant after considering all mitigations. Of these 80, only 22 of these involved mitigations (with the remainder having no associated mitigation). In other words, for 447 significant impacts that had associated mitigation measures, 425 were deemed not significant following mitigation, and 22 were still considered significant (a 19:1 ratio).

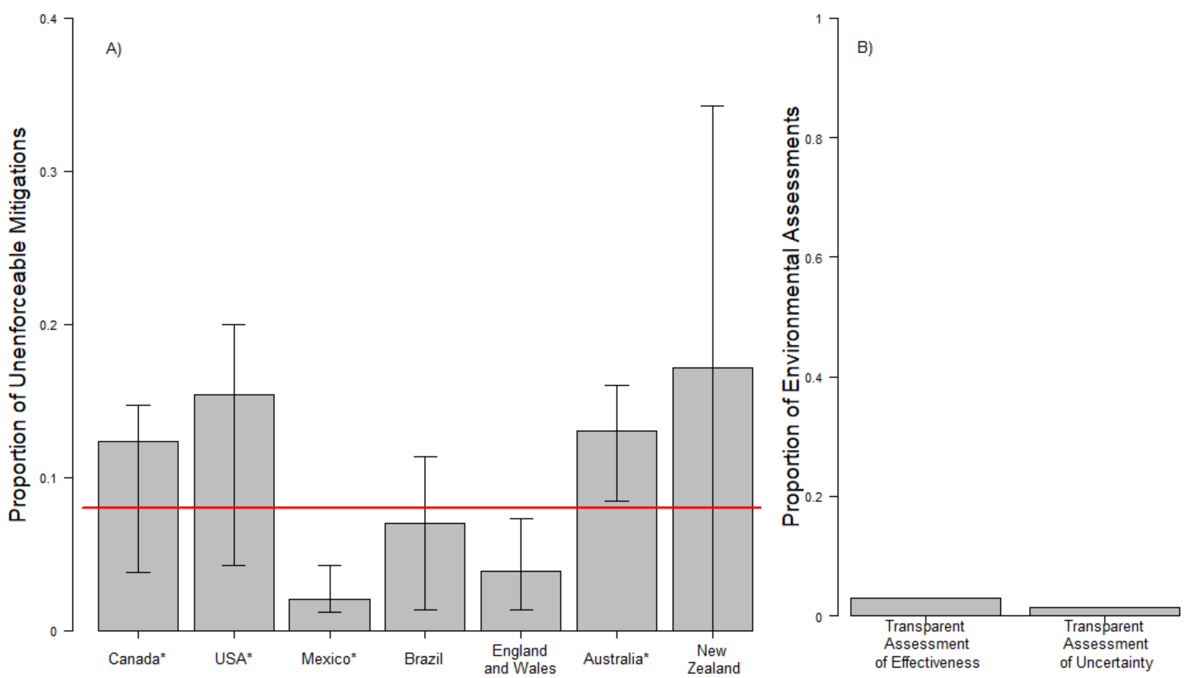

Figure 3. The proportion of (A) mitigation measures written in ambiguous and unenforceable language in each jurisdiction (bars represent 95% bootstrap CI of the median and red line represents global

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median) and (B) EISs in all jurisdictions that have explicit analysis of mitigation effectiveness and consider uncertainty of mitigation effectiveness. No single EIS considered both mitigation effectiveness and uncertainty. EISs were selected from a single state or province within the countries marked with an asterisk (*) along the X-axis.

Additionally, we found no EIS that assessed both mitigation effectiveness and uncertainty of impact reduction (Figure 3b). Rather, actions intended for mitigation were generally treated as effective despite research demonstrating the reverse (e.g. fish habitat compensation (Quigley and Harper 2006), or despite a lack of research into specific mitigation effectiveness (Duinker et al. 2012; Jacob et al. 2016). Furthermore, some mitigation proposals were worded in such a way that it was unclear if they would even be implemented, and were yet still considered effective. We found that 5-11% (bootstrap 95% CI of the median) of mitigation measures across jurisdictions were expressed in vague language that left ambiguous what actions, if any, would be taken (e.g. "where applicable, mitigation X will be installed"; "to the extent possible, mitigation X will be explored"; Figure 3). The consequence of this equivocal wording is that the developer's level of commitment to a given mitigation measure is unknown (Marshall 2002; Duinker et al. 2012; Lees et al. 2016).

Lastly, no EIS in our sample included additional mitigation measures to address cumulative impacts. Thus the number of potential cumulative impacts is equal to the number of residual cumulative impacts. As mitigations are central to EIAs, the lack of mitigations for cumulative impacts may reflect the fact that developers only have power to apply mitigations to the areas they have licences for, and larger scale mitigations would require additional work (Burris and Canter 1997; Piper 2001; Hellweg and Milà i Canals 2014). Indeed, the scope of EIAs are usually primarily focused on individual projects, which can limit the consideration of impacts from other projects and across supply-chains (Burris and Canter 1997; Duinker and Greig 2006).

In summary, the high confidence in mitigation measures expressed in EISs is questionable, because mitigation proposals in EISs are sometimes not enforceable (and in the case of cumulative impacts, not proposed) and often not scientifically verified (Hollick 1981; Duinker and Greig 2006; Duinker et al. 2012).

3.4 Likely Biased Significance Determination

Consultation with stakeholders other than developers is crucial for two reasons: (1) understanding what aspects of the environment are important for assessing impacts, and (2) determining significance where biophysical impacts have social or cultural implications (Canter and Canty 1993; Briggs and Hudson 2013; Ehrlich and Ross 2015). However, in all but one of our sampled EISs, stakeholders had no input in the determination of significance, and significance was instead determined by consultants (normally paid by project developers). In the outlier, a New Zealand EIS, a team of Maori stakeholders both determined the cultural values at risk and assessed impacts on these cultural values. In some EISs, local stakeholders were simply told of a planned development without being given the option to voice concerns (Figure 4). Most commonly, local stakeholder concerns were documented (with no follow-up) or responded to in facilitated meetings with no further opportunity to influence the design of the project or determine if their values were factored into significance determination (Figure 4).

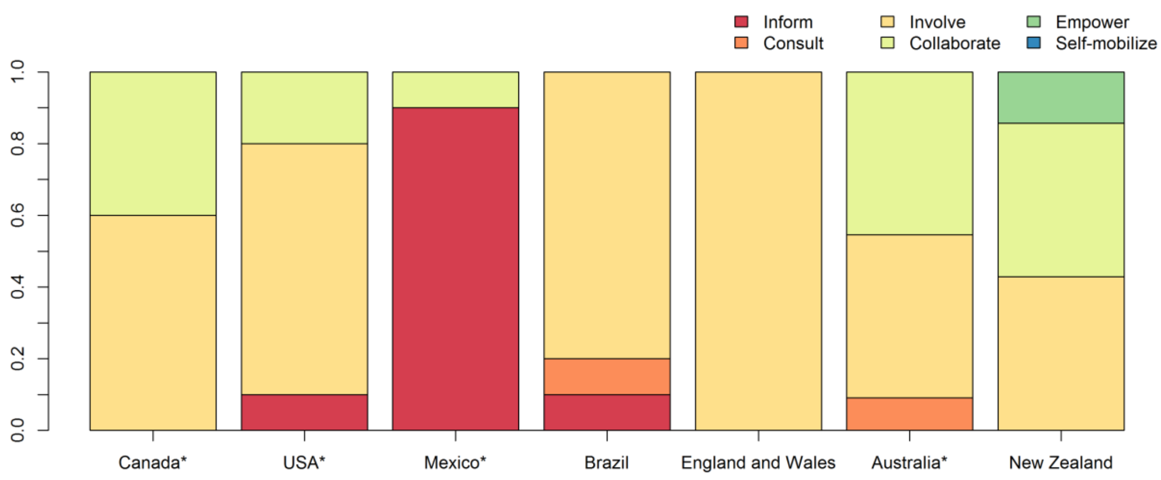

Figure 4. The proportion EISs from each jurisdiction that consulted stakeholders to various degrees. EISs were selected from a single state or province within the countries marked with an asterisk (*) along the X-axis. Refer to Table S4 for a description of the different categories of consultation in the legend.

In fact, certain stakeholder groups were not consulted at all in some EIAs. Community organizations were consulted in 51% of our sample, indigenous groups in 58% (exempting England and Wales), and environmental groups in 70%. Other groups were consulted more often: business and political groups were consulted in 88% and 97% of our sample, respectively. There are a few potential explanations for these disparities in representation. First, community, indigenous, and environmental groups may lack the capacity to represent their interests to the same extent as business or political groups. Second, these less represented groups may not have elected to participate in the consultation process as much as the more represented groups, for various reasons including not having a stake in proposed development sites (however, in the two jurisdictions where there are strong legal requirements for First Nation consultation—Canada and New Zealand—we found these groups were consulted in 100% of our sample). Finally, consultation of relevant stakeholders may have disproportionately failed to include environmental, indigenous, and community groups even when these groups had a stake in a proposed development. Our findings cannot distinguish among these explanations, and invite further research on this gap in consultation. The literature documents case studies of EIAs suppressing concerns of local groups and those who might be against development (O'Faircheallaigh 2010) . Limited consultation with indigenous groups has extra consequence, as indigenous groups often have dependencies on and histories linked to the environment not shared by others (Stevenson 1996; Banerjee 2000). EIAs may thus exacerbate a power imbalance in environmental decisions that has contributed to cultural loss for indigenous people worldwide (Banerjee 2000; Ward 2001; Cashmore and Axelsson 2013).

Though quantitative thresholds were sometimes factors in determining the significance of an impact (48% of our sample used quantitative thresholds for a subset of impacts, and 42% did so for a subset of cumulative impacts), we found that every EIS relied on the consultants' judgement for the majority, if not all, determinations of impact significance. While using professional judgement is itself not cause for concern, relying on professional judgement without clearly outlining the considerations that influence significance determination lacks transparency (Jones and Morrison-Saunders 2016). Based on our sample, 69% of EISs did not clearly document the methods used to determine significance, and for the 31% that did, significance was based on ambiguous qualitative criteria with little explicit information on how these were derived or applied. For example, significance was often defined as being dependent on the sensitivity of the environment to the impact and the magnitude of the impact, without outlining how one or either of these inputs was determined. Furthermore, professional judgement acquired without a structured protocol to counteract cognitive biases and overconfidence in assessment is prone to provide misleading results (Morgan 2014), and we found no EIS that outlined any protocol used to elicit professional judgements.

As mentioned previously, across the jurisdictions we investigated, developers pay the consultants who prepare EISs. Finding few (or no) significant impacts often fulfills the financial interest of both the developer and the consultant, and can go towards maintaining a strong business relationships between the two (Hollick 1984). While we cannot measure how this system affects EIS conclusions drawn, this EIS practice may serve to normalize a bias due to conflict of interest (Hollick 1984). Judgements are easily influenced by affiliation with interested partisans (Moore and Loewenstein 2004; Moore et al. 2010). Conflict of interest may bias consultants to conduct the main components of an EIS (scoping, assessing mitigation, conducting consultation, and determining significance) in ways that are favourable to the developer. In general, conflict of interest may bias consultants to present the environmental impacts of a project as negligible to a decision-maker, minimizing the chances of identifying trade-offs between economic development and environmental quality and therefore presenting a case whereby avoiding economic benefits from development is considered an unnecessary loss. Jurisdictions with processes designed to avoid conflict of interest (such as in the Netherlands, where an independent body of experts review each EIS) may not follow the patterns we found, but future research is required to determine if this is the case. Our aim is not to accuse consultants of dishonesty or incompetence; however, we point out that the potential institutional bias introduced by this conflict of interest is problematic (Moore et al. 2010).

4. Conclusions

Our findings suggest that in the seven jurisdictions we address, EISs often contain questionable analysis and lack transparency, which may bias their conclusions against determinations of significant negative impacts. While there are other regulatory processes and considerations that affect final decisions, EISs ostensibly give scientific credibility for decisions, so sound research practices are important. Improving EIS practices will require addressing the problems we have outlined, from scoping and impact prediction to public participation and significance determination (Morgan 2012).

Six major changes could help to improve EISs' utility as legitimate science-based resources for environmental decision-making:

1. The spatial and temporal scope of assessments should be ecologically justifiable and explicitly consider cumulative impacts, or explicitly link to larger-scale Strategic Environmental Assessments, encompassing the ranges of ecosystem components affected and the duration of demonstrated lag impacts from relevant literatures (Shepherd and Ortolano 1996; Duinker et al. 2012; Bidstrup et al. 2016).
2. Interactions among impacts should be explicitly considered and in reference to available evidence, acknowledging evidence that interactive, non-additive effects are the norm (Crain et al. 2008).
3. Mitigation actions should be stated in ways that are enforceable. The degree of effectiveness of all mitigations should be evaluated, with uncertainty acknowledged, and contingencies considered for potential mitigation failure. Conversely, an impact should not be considered successfully mitigated, and thus not significant, unless planned mitigations have a demonstrated effectiveness in appropriate contexts (Hollick 1981; Duinker et al. 2012).
4. Stakeholders should have input into impact significance determination whenever impacts may have local, social or cultural consequences – likely the majority of cases (O'Faircheallaigh 2010).
5. Policies should force developers to comply with changes 1-4 listed above by providing regulators with the technical and personnel capacity to appropriately assess scientific rigour in order to approve or reject the EIS based on assessment quality (Clark 1999; Kirchhoff 2006), and make environmental audits compulsory to ensure developer's compliance with mitigation commitments.

6. The inherent conflict of interest in EIS authorship must be eliminated (e.g. by having developers pay into a common fund, administered by governments, to retain independent experts to author or review EISs) (Hollick 1984; Moore et al. 2010).

To be a truly transparent and robust tool of environmental protection, EIA needs to embrace current evidence and practices when relying on research. Failure to improve regulation and practice allows EISs to obscure and facilitate important environmental impacts more often than they reveal and prevent them.

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