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RESEARCH ARTICLE

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# Ecological stakeholder analogs as intermediaries between freshwater biodiversity conservation and sustainable water management

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## Abstract

Water security is essential for human well-being and is among the biggest challenges in environmental governance. Governments and nonprofit organizations alike are gaining increased appreciation for the contributions of intact ecosystems to water security, whereas conservation scientists call for decisive action to address the dire condition of earth's freshwater ecosystems and biodiversity. Stakeholder-based, Habermasian decision-making frameworks such as integrated water resources management (IWRM) are widely used to equitably manage complex water systems, and ecologists have developed increasingly sophisticated frameworks (e.g., environmental flows) to quantify and anticipate the ecological outcomes of water management decisions. IWRM implementation is criticized for being excessively top-down whereas ecological frameworks in water decision-making can fail to account for the cultural and societal values of ecosystems, and it remains unclear how best to connect the desired bottom-up implementation of IWRM with the expert-based, top-down structure of hydro-ecological research. We revisit and elaborate upon the ecological stakeholder analog (ESA) concept, which treats ecological phenomena (e.g., species and processes) as stakeholders and ecological information as interests and positions with respect to water management. We then illustrate how ESAs can address the many calls to improve environmental flows and IWRM strategies by improving their integration, and how established conceptual frameworks from stakeholder theory applies readily to ecological stakeholders.

**KEYWORDS**

ecosystem services, environmental flows, freshwater biodiversity, IWRM, sustainable development goals, water diplomacy, water policy, water security

## 1 | INTRODUCTION

The quantity, quality, and availability of fresh water are essential to human enterprise and well-being, and water security is among the most important and difficult challenges of environmental governance at multiple spatial scales (Pahl-Wostl, Palmer, & Richards, 2013). Water supply-related crises are ranked among the Top 3 global risks to human societies, behind only weapons of mass destruction and extreme weather events (World Economic Forum, 2017). Freshwater resources management is key to achieving the United Nations sustainable development goals (Irvine, 2018; United Nations, 2018; Vörösmarty et al., 2018), and it is becoming increasingly important in the face of climate change (Schewe et al., 2014). Meanwhile, nature's role in water security is being increasingly recognized (Harrison et al., 2016), with beneficial effects (ecosystem services) of freshwater ecosystems valued at US$4 trillion annually (Béné et al., 2016; Costanza et al., 2014). The European Water Framework Directive, among the most ambitious and influential international environmental agreements, includes ecological objectives as one of the primary goals for freshwater resources management (European Communities, 2000). Nature-based solutions are considered a significant part of the global freshwater development toolbox (Cohen-Shacham, Walters, Janzen, & Maginnis, 2016; United Nations World Water Assessment Programme, 2018), and prevailing opinion on the connection between biodiversity and water security is clear from high-level panels of water resources experts (Vörösmarty et al., 2018).

Fresh water, amounting to 0.01% of water on the planet (Gleick, 1996), supports nearly 10% of all known species, including 30% of vertebrates (Mittermeier, Farrell, Harrison, Upgren, & Brooks, 2010). Unfortunately, the very ecosystems that contribute so substantially to global water security and biodiversity are perhaps the most threatened in the world (Dudgeon et al., 2006; Ricciardi & Rasmussen, 1999). Populations of many freshwater vertebrates have shown declines of >80% over the last 50 years (World Wildlife Fund, 2016). Given that these disproportionately threatened systems have also been neglected in research, funding, and conservation action (Darwall et al., 2018), freshwater ecosystems are now placed among the Top 3 global conservation priorities (Jucker et al., 2018). The improvement of water policy and management frameworks is a major priority in protecting freshwater life (Dudley, Harrison, Kettunen, Madgwick, & Mauerhofer, 2016; Irvine, 2018). Indeed, a recent global meta-analysis showed that poor governance structure was the strongest predictor of declines in waterbird populations, a proxy for freshwater biodiversity (Amano et al., 2018). Better governance and management of freshwater resources could thus greatly benefit biodiversity conservation in addition to achieving sustainable development goals (Darwall et al., 2018; Karlsson-Vinkhuyzen et al., 2018).

The vulnerability and interconnectedness of freshwater ecosystems and the ubiquitous consumption of freshwater resources by human societies inevitably lead to conflict, and trade-offs between ecological and societal needs are a necessary and complex reality (Dudgeon et al., 2006). This requires management that is flexible, adaptive, reconciliatory, and often carried out at the basin scale (Chen & Olden, 2017; Dudgeon et al., 2006; Hermoso, Cattarino, Kennard, Watts, & Linke, 2015). Researchers from both the ecological and sociopolitical sciences have developed powerful frameworks for addressing this difficult management context. Integrated water resources management (IWRM) is the dominant paradigm for freshwater resources management from a governance perspective, being implemented in over 100 countries (Allouche, 2016; Mollinga, Dixit, & Athukorala, 2006; Smith & Clausen, 2017). This consensus-based approach draws from Habermas' (1984) concept of communicative rationality and the use of collaborative planning and network power (*sensu* Innes & Booher, 2010) to create governance mechanisms that are resilient to complex, dynamic systems. Among ecologists and conservation scientists, research on environmental flows (e-flows) has yielded increasingly advanced scientific approaches to understanding and controlling the impacts of water management on freshwater biodiversity in riparian systems (e.g., Arthington, Kennen, Stein, & Webb, 2018; Poff et al., 2010). The growing sophistication of both approaches to navigating water management presents an unresolved challenge to integrate these two dependent but not yet fully connected fields for effective implementation.

The urgency of issues at the intersection of water security and freshwater biodiversity conservation necessitates a way of integrating the rich and technical information provided by ecological science with the participatory, bottom-up, highly reconciliatory nature of real-world water management. This is recognized by authors in both fields, with IWRM researchers criticizing overly technocratic approaches yet calling for expert ecological information (Smith & Clausen, 2017) and e-flows (i.e., environmental flows; maintaining streamflow characteristics to support ecological objectives) researchers advocating improvements in management decision-making, particularly with respect to societal variables such as the cultural and heritage value of biodiversity (Arthington et al., 2018; Jackson, 2017; Poff, 2018). In this paper, we highlight and elaborate upon a recently developed framework that bridges this divide, facilitating freshwater management at the nexus of water security and freshwater biodiversity conservation. In particular, we seek to emphasize its utility in reconciling and connecting contemporary paradigms from societal and natural domains (*sensu* Islam & Susskind, 2012) and clarify key concepts in its structure and implementation.

## 2 | THE ECOLOGICAL STAKEHOLDER ANALOG CONCEPT AND ITS ROLE WITH RESPECT TO CONTEMPORARY MANAGEMENT PARADIGMS

van Rees and Reed (2015) developed the concept of “Ecological Surrogate Stakeholders” (although they are more accurately called ecological stakeholder analogs [ESAs], which we will use throughout) as a way of improving integration of ecological information into stakeholder-based water management frameworks such as IWRM and Water Diplomacy (Islam & Susskind, 2012). For the sake of brevity, we refer readers to the original publication for a thorough treatment of the concept, and only briefly recapitulate key terms and ideas in this section. An ESA is an ecological phenomenon (e.g., a population, species, local assemblage, community, ecosystem, or biological index) that is affected in some way by a water management decision and treated analogously to an actual (human or societal) stakeholder in the decision-making process. Like conventional stakeholders, these analogs have positions with respect to a management decision (dictated by, e.g., flow-response relationships, Arthington, Bunn, Poff, & Naiman, 2006) that are attributed value based on their relation to other stakeholders. ESA interests (the ecological processes and mechanisms underlying and driving flow-response relationships) give rise to these positions when their response to a management decision is combined with other stakeholders' perceived value of that response. For example, the observed response of rainbow trout *Oncorhynchus mykiss* to reduced streamflow can be reduced growth rate (Harvey, Nakamoto, & White, 2006), which can have population-level impacts because body size limits reproductive output and survival. ESA positions can be viewed as positive or negative according to either anthropocentric or intrinsic value ethics of the stakeholders and decision-makers involved (Figure 1). In the case of rainbow trout, lower population size and smaller fish may be a highly undesirable impact for recreational or commercial anglers, thus attributing a negative value to the observable position that fewer, smaller fish appear when streamflow is reduced.

van Rees and Reed (2015) argued that approaching ecological phenomena as ESA improves water management by (a) encouraging the use of hypothesis-based, mechanistic ecological research rather than purely descriptive studies and (b) facilitating the detection and adoption of mutual gains ("win-win") management solutions. These two advantages are especially relevant to current trends in improving ecological water management, including a desire for more process- and rate-based metrics (Arthington et al., 2018; Poff, 2018), emphatic calls to make better use of nature-based solutions (Cohen-Shacham et al., 2016; United Nations World Water Assessment Programme, 2018), and a need for policy solutions that simultaneously achieve wildlife conservation and water security objectives (Harrison et al., 2018). In addition to these key advantages, we add that the ESA concept might also help achieve separate goals for the improvement of water management from both a governance and an ecological perspective. By bringing ecological phenomena "to the table" with stakeholders in a basin, facilitators reduce the typically top-down and technocratic aspect of ecological water management by clearly and objectively presenting the behavior of an ESA with respect to management decisions, rather than restricting the decision-space with inflexible, difficult to understand, and pre-defined boundaries. This could greatly increase the transparency, public legitimacy, and repeatability of this challenging aspect of water management, addressing current concerns over the implementation of IWRM (Smith & Clausen, 2017). This also enables the integration of high-level ecological science into a bottom-up approach to water management, encouraging stakeholder interest in asking relevant and testable ecological questions, and facilitating a joint-fact-finding approach that builds trust among otherwise conflicting stakeholder groups (Islam & Susskind, 2012). By assessing the salience and relative priority (see below) of ESA through stakeholder analysis, water managers could also better direct research to focus on the ecological phenomena and particular questions that are most relevant to the satisfaction of multiple stakeholder interests in the basin. Freshwater conservation necessitates basin-specific solutions (Dudgeon et al., 2006), and implementing ESA may encourage approaching management solutions at the basin scale, taking into account local biodiversity, hydrology, and socioeconomic characteristics. At the same time, including ESA allows for the integration of other, noneconomic values of freshwater biodiversity, which are increasingly viewed as a priority for improving the equity and efficacy of environmental flows research (Arthington et al., 2018; Noble, Fulton, & Pittock, 2018). This contributes to current interest in improving the integration of community-based cultural values in water and ecosystem management (Daniel et al., 2012; Langhans et al., 2018).

Before the ESA concept can be implemented, however, significant conceptual clarification and elaboration are needed. Van Rees and Reed's (2015) introduction of the topic raised questions about how ESA might be selected or prioritized, as well as philosophical and practical concerns about the nature of the framework. In the following sections, we expand upon and further develop the ESA concept and clarify its compatibility with established stakeholder-based approaches

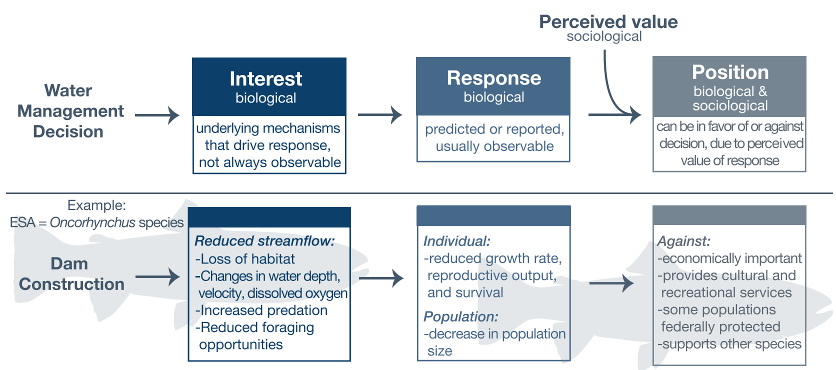


**FIGURE 1** The position of an ESA, an integral component of stakeholder negotiation, is the result of the biological response to a water management decision combined with the perceived value (anthropocentric or intrinsic) of that response. Observed responses are produced by underlying biological or ecological mechanisms, which are treated as the interests of an ESA. In this example, dam construction (a water management decision) reduces streamflow (interest), which decreases reproductive output, survival, and population size (responses) of an *Oncorhynchus* species, the ESA. Stakeholders, such as those that benefit culturally, recreationally, or economically from a healthy trout population, would likely be against (position) the decision, based on their values. Example adapted from Harvey et al. (2006). ESA, ecological stakeholder analog [Colour figure can be viewed at wileyonlinelibrary.com]

to increase its utility and encourage its use in addressing global water security and the ongoing freshwater biodiversity crisis.

## 3 | IDENTIFICATION OF ESAS

The definition of a stakeholder, which began as "[a]ny group or individual who can affect or is affected by the achievement of the organization's objectives" (Freeman, 1984), has been revisited repeatedly over the last 30 years (Bryson, 2004; Mitchell, Agle, & Wood, 1997; Wagner Mainardes, Alves, & Raposo, 2011). The literature of stakeholder theory provides guidelines for managers to understand which parties should be included and consulted for management decisions (Neville, Bell, & Whitwell, 2011) and has focused primarily on the identification and prioritization of stakeholders. Here, we show that existing stakeholder concepts can be applied to ESAs. Mitchell et al.'s (1997) foundational work outlined three criteria for classifying stakeholders: power, legitimacy, and urgency. The degree to which each of these characteristics apply to a particular stakeholder can be readily used to identify and prioritize ESA (Figure 2). We explain the nature of each of these characteristics below and clarify how they might apply to a typical ecological stakeholder.

### 3.1 | Power

Power refers to a stakeholder's ability to influence other stakeholders' interests or management decision outcomes (Blau, 1964; Freeman, 1984; Weber, 1947). Power can manifest as the potential for force through physical or legal action (coercive power), for provision of goods or services (utilitarian power) or for social authority or popularity sufficient to create cultural pressure on the other entities (normative power; Mitchell et al., 1997).

Because ESA lack political agency, their power is indirect and derived from anthropogenic value attributed to their position, or response, to proposed management. Power in ESA can be derived from a species' economic value, political or cultural importance, or legal status. For example, trout (*Oncorhynchus* spp.) in the American Northwest have substantial power as an ESA due to their economic and cultural importance for fishing, leading to their explicit consideration in water management (Gosnell, Haggerty, & Byorth, 2007). Indeed, legal protection alone can give power to ecological phenomena, especially for species with listed or endangered status. For example, a single nest of Say's phoebe (*Sayornis saya*, federally protected under the Migratory Bird Treaty Act) in California delayed repairs on a highway overpass for weeks while the birds completed their nesting cycle (Ghori, 2015).

### 3.2 | Legitimacy

Legitimacy refers to how positions and interests of stakeholders are viewed in the context of the cultural norms and expectations created by society (Mitchell et al., 1997). In short, legitimacy pertains to how much a stakeholder is thought to deserve a role in the decision-making process. Suchman (1995) distinguished between three forms of legitimacy: moral (cultural norms), pragmatic (the instrumental importance of the stakeholder to the decision system), and cognitive (common assumptions or beliefs that are taken for granted). ESA can possess any of these three types of legitimacy. Many native species typically have high moral legitimacy, because much of society feels that they have intrinsic value or a right to exist (Driscoll, Wiebe, & Dyck, 2012). Ecological phenomena gain increasing legitimacy given evidence or past experience that they can be negatively impacted by management decisions. For instance, migrating fish species, such as Chinook salmon (*Oncorhynchus tshawytscha*), have high moral legitimacy as ESA in dam management decisions within their native range because increased juvenile mortality has been caused by previous dam constructions (Raymond, 1979). Delayed migration during low-flow years and increased exposure to dissolved gases during high-flow years were among the causes (i.e., the interest analogs) of juvenile mortality resulting from dam construction along the Snake River and led to record lows of returning adults in subsequent years (Raymond, 1979). Recent work on an endemic crayfish in Australia gives a strong example of how legitimacy increases salience for native species (Noble et al., 2018). By contrast, widely introduced species such as the Louisiana swamp crayfish (*Procamburus clarkii*), which has become invasive in the Mediterranean (Scalici & Gherardi, 2007), have low moral legitimacy but may be attributed pragmatic legitimacy due to their negative impacts on other ESA or desired stakeholder outcomes.

### 3.3 | Urgency

Mitchell et al. (1997) define urgency based on the magnitude and/or immediacy of the impact that a decision may have on the stakeholder. Stakeholders with high urgency typically gain attention from water managers because their interests are strongly affected by

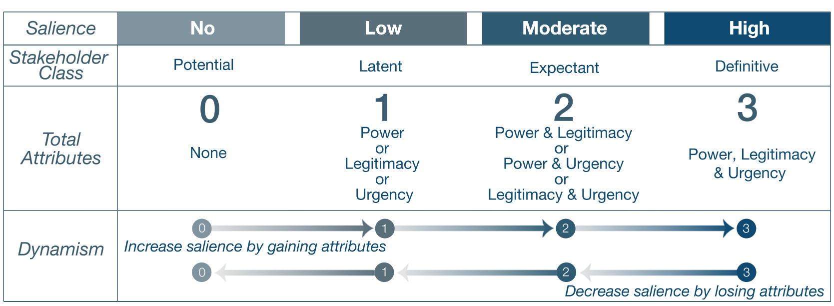


**FIGURE 2** The total number of attributes (power, legitimacy, and urgency) a stakeholder possesses informs its salience. For example, stakeholders possessing all three attributes are classified as "definitive stakeholders" and those with none as "potential stakeholders." Salience is dynamic, and stakeholders may gain or lose attributes at any time. Adapted from Mitchell et al. (1997) [Colour figure can be viewed at wileyonlinelibrary.com]

management decisions or because their interests are directly violated by a lack of action (i.e., the status quo). Urgency for ESA is straightforward when the potential impacts on freshwater biodiversity are apparent and severe (e.g., extirpation, significant declines in function). Ecological factors already impacted by current conditions in a watershed have higher urgency than do those for which impacts are predicted or hypothetical.

The concept of temporal urgency, however, can be problematic for ESA because ecological phenomena and their responses to environmental change often operate at time scales that differ significantly (both shorter and longer) from those of human systems. Extinction debt, where the ecological impacts of a disturbance are not evident until many years later, is a key manifestation of such a mismatch of temporal scale. For example, habitat loss and fragmentation in Europe resulted in delayed extirpations of forest plant species up to a century after forest clearing (Vellend et al., 2006). Consequently, if temporal scale is unaccounted for, ESA that suffer lagged harm from contemporary water management decisions would have low apparent urgency, thus reducing stakeholder salience (see below) despite the existence of a very real threat. Driscoll and Starik (2004) cautioned that decision-makers should evaluate management actions with respect to “current and future generations and both the short- and long-term impacts of decisions on the natural environment” (p. 62). Urgency within the ESA framework therefore refers to the magnitude of harm across a temporal scale that effectively captures the position of the ESA in question.

## 4 | STAKEHOLDER PRIORITIZATION

Stakeholder identification and prioritization is driven by salience (*sensu* Mitchell et al., 1997). Although stakeholders can be identified by the possession of any of the criteria outlined above, priority takes into account the degree of legitimacy, power, and urgency possessed by a given stakeholder or stakeholder group. Mitchell et al. (1997) identified eight classes of stakeholders based on which and how many stakeholder identification attributes they possess. Stakeholders with only one attribute are referred to as latent stakeholders, those with two attributes are expectant stakeholders, and those with all three attributes are highly salient or definitive stakeholders (Figure 2). These stakeholder classes are further subdivided by the combination of attributes they possess; Mitchell et al. (1997) describes these in detail, so for the sake of brevity, we refer readers to that text and describe only a subset that are particularly relevant to ESA.

Dependent stakeholders are those possessing legitimacy and urgency without direct power. In the context of ecological stakeholder analogs, this might include ecological factors that lack legal protection and economic value but that are strongly affected by the management decision in question and with high legitimacy for its presence in the system. These stakeholders must rely on other, more powerful stakeholders (e.g., dominant stakeholders) to represent their interests. In referencing the Exxon Valdez oil spill, Starik (1993) identified ecological factors such as marine mammals and the local ecosystem as dependent stakeholders.

Dominant stakeholders have both power and legitimacy but lack urgency in their claims. ESA that provide ecosystem services, are protected by law, or are of cultural or economic importance but will not be immediately or critically impacted by a management decision fall into this category. Dominant societal stakeholders in environmental conflicts are often private corporations and businesses or governmental or nonprofit organizations whose interests (positively or negatively) affect the influence and positions of dependent ESA. If dependent or dominant stakeholders acquire the missing attribute, they can transition to definitive stakeholders. The Murray crayfish (*Euastacus armatus*), for example, possesses economic and legal power, legitimacy as a culturally important keystone species, and urgency given dramatic declines in their abundance across their geographic range (Noble et al., 2018). Classifying ESA in a manner analogous to societal stakeholders facilitates their prioritization both relative to each other and to other stakeholders in water management decision-making.

Beyond the different dimensions of salience intrinsic to a given stakeholder, stakeholders and their ecological analogs can also derive importance and priority from their relationships to other stakeholders. A stakeholder-issue interrelationship diagram is often created to place stakeholders and issues of interest in context using a conceptual map of their relationships with other stakeholders (Bryson, 2004). This map (also called a graph, *sensu* West, 2001) can include the directionality of relationships using arrows and can be weighted to depict the strength of the relationship between stakeholders. Ecologists will be familiar with this type of figure because a similar structure is used to depict trophic or food webs (e.g., Pimm, Lawton, & Cohen, 1991). In a food web, arrows depict the flows of energy that connect ecological factors in a system. Although a food web is a simplified view (i.e., a model) of a complex system, it yields insights on system structure and vulnerability, as well as a species importance in the community (e.g., node centrality, when treating species or functional groups as nodes).

The same network metrics that are used to analyze and evaluate food webs can be used to analyze stakeholder networks via stakeholder-issue interrelationship diagrams. Including ESA in this context involves integrating an existing food web and stakeholder-issue interrelationship diagram by showing the connections between societal and ecological stakeholders. Two types of network models can be constructed using ecological factors, each serving a distinct purpose: (a) ecological networks of phenomena such as species, ecosystems, and services, which allow prioritization and selection of ESAs based on their relationships to one another, and (b) integrated stakeholder-issue interrelationship diagrams, in which ESA are related to actual stakeholders to better understand how their interests and positions in the water management issue align with those of other parties (Figure 3). The first of these contributes to basin-specific planning and prioritization for the collection of ecological information, allowing for ecologically informed selection of what biotic variables might be the most important to monitor or predict. The results can

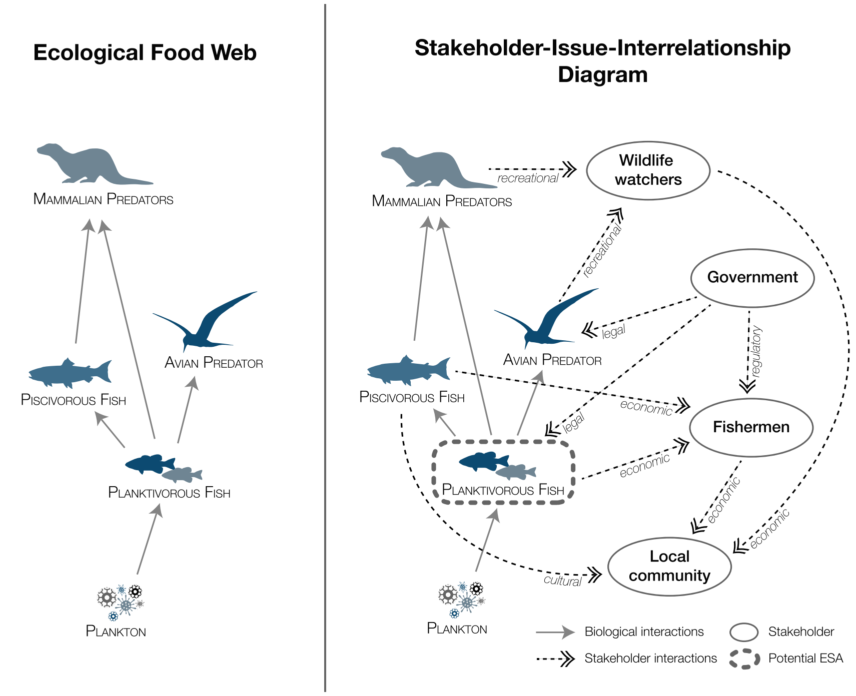


**FIGURE 3** Left: A simple ecological food web depicting energy flow (via ingestion) in an aquatic system. Right: The same food web integrated with a stakeholder-issue interrelationship diagram. Stakeholders and are in closed circles, and a potential ecological stakeholder analog (ESA), planktivorous fish, is identified in a dashed rectangle. Gray arrows represent biological interactions (energy flow) and dashed arrows represent stakeholder interactions (e.g., economic or cultural value, or legal protection). Arrows connect stakeholders with the environment and with each other. The chosen ESA in this case is the most connected ecological factor in this system, exhibiting direct interactions with all other ecological factors in addition to holding legal status from the government and economic value for fisherman. For simplicity, only one ecological factor was chosen as an ESA, but any number of ecological stakeholders can be involved in a water management system if their levels of salience justify inclusion. [Colour figure can be viewed at wileyonlinelibrary.com]

be integrated into the second type of network model, which clarifies the broader interactions between various ecological phenomena and important societal entities, integrating community values and other social variables into water management decision-making.

Several authors have asserted that stakeholder priority may be better assessed with respect to coalitions or groups of stakeholders (Neville et al., 2011; Neville & Menguc, 2006). The explicit mapping of relationships between ESAs using a network-based approach facilitates such an approach and can be based on ecological similarities (e.g., habitat use and feeding guild), trophic interactions, or taxonomic relatedness. Using such a grouping approach for ESA would facilitate research interest in broadening the use of indicators in environmental flows to community- or ecosystem-based metrics (Arthington et al., 2018). Medema et al. (2017) emphasized the importance of bridging organizations, which are stakeholder groups that unite suites of stakeholders according to some common interest, even if this interest is outside of the issue under decision. In the context of societal stakeholders, bridging organizations are often created especially for facilitating collaboration and social ties between stakeholders to simplify the decision-making process and increase acceptance of outcomes. This also lends itself to network depiction (where groups of organizations can be treated separately or as a single node). Examining ecological webs and interactions between ecological factors or ESA takes advantage of existing ecological knowledge to better understand the interconnectedness of the stakeholder network at a larger scale and reduces the complex task of prioritization to a potentially more repeatable and transparent process. Resulting conceptual models of the interactions between ecological phenomena, societal stakeholders, and management decisions are also readily compatible with many environmental decision-making frameworks and software packages (e.g., Zonation, Moilanen, Kujala, & Leathwick, 2009; see Creech, McClure, & van Rees, 2018 for an implementation example in a freshwater system).

In addition to selecting and prioritizing ESAs based on economic importance and social values, a number of well-established concepts in ecology can be employed to take advantage of the interconnections among ecological factors (i.e., potential ESA), in the same way, decision-makers would normally approach conventional stakeholders. Important ecological phenomena, such as umbrella species (Lambeck, 1997), keystone species (Mills, Soulé, & Doak, 1993; including ecosystem engineers; Jones, Lawton, & Shachak, 1994), flagship species (e.g., Dietz, Dietz, & Nagagata, 1994), bioindicators (e.g., Bourgoin, 1990), and surrogate species (a more general term that covers a variety of proxy relationships that would need specification; Caro, 2010), all make effective ESA based on their established relationships and correlations with other ecological phenomena. If their ecological positions and interests are similar or aligned, combining ecological factors simplifies analysis and decision-making by reducing the number of stakeholders necessary to capture the ecological dynamics important for a management problem. For example, some species, such as ecosystem engineers, directly affect a contingent of other ecological factors (e.g., beavers *Castor canadensis* build dams, thus creating habitat for lentic wetland specialists). These ESA gain increased salience because their analog-positions or -interests represent a number of other stakeholders.

It is also important to note that criteria of salience are dynamic, so a given ESA can alternately gain and lose power, legitimacy, or urgency throughout a decision-making process. For example, increased media coverage might grant normative power to an ecological stakeholder, whereas newly published population projections might increase or decrease urgency or legitimacy. Accordingly, stakeholders can switch between different levels of salience and classification based on the number of criteria they possess, and the relative influence of those criteria on other stakeholders in the system (Figure 2). For example, ESA with high power (e.g., species given legal protections) may be highly salient by that single criterion. We do acknowledge, however, that legal listing is typically associated with one or more other attributes (e.g., urgency necessary for listing and social legitimacy due to listing), and the contribution of these attributes also increases salience.

By matching established ecological theory to concepts of stakeholder salience and prioritization, the ESA concept allows for a more comprehensive analysis of the risks and opportunities for ecologically informed water management and reduces the risk of system oversimplification. The techniques and criteria for prioritizing and grouping specific ecological factors can be easily integrated into identification, prioritization, and grouping of ESA, thereby promoting further inclusion of key ecological concepts and information in water management. In the next section, we address potential concerns regarding the ESA approach and highlight key areas for further research to enhance its utility.

## 5 | PRACTICAL CONCERNS AND PRIORITIES FOR FURTHER RESEARCH

In order to implement the ESA concept in stakeholder-based approaches to water management, a number of practical concerns must be addressed to guide effective use. One potential concern is whether giving ecological phenomena a proverbial “seat at the table” leaves them vulnerable to greater exploitation than would simple top-down environmental regulation, opening the door to decisions that undermine the value or integrity of those phenomena. We believe that this concern arises from a misinterpretation of the concept, in which stakeholder-based decision-making involving ESAs is viewed as an alternative to existing top-down controls. This is not the case; we view the ESA concept as being added to the existing protections for wildlife and ecosystems, and one that operates within existing decision-making frameworks that attribute value to ecological objectives (e.g., IWRM and ecosystem-based management and multicriteria decision analysis; Langhans et al., 2018). It might be true that, in some cases, placing an ecological phenomenon as an equal part of a stakeholder negotiations process could lead to some disenfranchisement, but in a well-orchestrated stakeholder co-management process, this should only happen where such phenomena have no preexisting legal protections, affect no societal or ecological stakeholders' interests in any way, are not the target of any conservation efforts or interest from other stakeholders, and whose claim has none of the aspects of salience explained above (power, legitimacy, nor urgency). In such cases, it is perhaps defensible that the phenomenon in question has no legitimate or convincing claim to influence the water management decision. This worst-case scenario is still preferable to the persistent undervaluation of ecological phenomena in most current freshwater management systems (Harrison et al., 2018; Vörösmarty et al., 2018) and at least has some transparent, traceable justification for disenfranchisement.

A similar objection might be that basing the interpretation of ESA positions on value attributed by societal stakeholders leads to a commodification of nature, which excludes noninstrumental evaluations of ecological phenomena. On the contrary, the concept does not depend on any particular type of value but instead generates position analogs and stakeholder salience from intrinsic, instrumental, cultural, or other sources of value attributed to the ecological phenomenon. Given that conservation efforts are already limited by what humans perceive as valuable (e.g., Sutherland et al., 2018; Williams et al., 2018), this aspect of the concept creates no new problems.

Another potential concern is that, even if they are being treated as stakeholders, ESA cannot actually represent themselves, and their position and interest analogs must be communicated by a societal entity to be part of co-management. The question of who gets to speak for the fish, as it were, becomes a very important one, considering potential biases and abuse of power by a chosen representative. We suggest that ecological scientists have a clearly delineated role as honest brokers (*sensu* Pielke, 2007) in the implementation of the ESA concept, wherein they provide objective information with quantified uncertainty on, for example, flow-response behavior (part of the analog position) and the responsible mechanisms (the analog interest). This not only allows scientists to restrict their responsibilities to tasks for which they are professionally trained but also encourages joint fact finding, in which potentially opposing societal stakeholders are both invested in the process of gathering scientific information to inform management, typically through a neutral third party. In such circumstances, paying for a scientist's time should be done in such a way as to prevent actual or apparent conflicts of interest. Where IWRM processes such as co-management and stakeholder negotiations highlight a need for greater information on an ESA's interest or position, this can direct ongoing scientific research in the basin and give stakeholders a sense of ownership and participation in data collection, an otherwise highly alienating stage of the decision-making process.

Although this paper provides significant clarification and depth to the concept of ESAs, the resolution of presently unaddressed questions, both practical and philosophical, would greatly benefit the contribution of this concept to uniting current efforts toward improving global water security and safeguarding freshwater biodiversity. No formal protocol has been developed for implementing ESA nor has the concept been tested in any water management scenarios (although opportunities may be arising in some study systems; van Rees, 2018). Further guidelines on how to select effective ESAs are needed, and a better understanding of the relationships between hydrological changes and freshwater biodiversity—especially outside of riparian systems, to which most research has been restricted—will be essential for dictating ESA positions and interests where time or funding is unavailable for empirical research (Rolls et al., 2018).

## 6 | CONCLUSION

We synthesized research from the ecology, wildlife conservation, business management, environmental governance, and water resources literature to provide clarification and elaboration on ESAs, a concept that bridges the divide between sophisticated, expert-derived ecological information and stakeholder-based co-management strategies for freshwater resources. In particular, we explain the potential role of ESA in modern IWRM and demonstrate how the basic principles of stakeholder identification and prioritization are directly applicable to ecological phenomena. ESA can be treated analogously to societal stakeholders, allowing the integration of important concepts from ecology and conservation, including trophic interaction webs, indicator species, and flow-response relationships, with existing stakeholder-based management approaches. This integration also allows the clearer attribution of a diversity of noneconomic cultural and societal values to ecological phenomena.

Framing ecological factors as ESA may improve stakeholder interpretation of ecological information and simplify the decision-making process, thereby increasing the trust in, and credibility of, water resources co-management. The ESA concept additionally addresses calls for improvement from both the ecological and social sides of water resources management, making technical ecological information compatible with the highly participatory, negotiations-based, "bazaar" environment (Lankford & Hepworth, 2010; Smith & Clausen, 2017) envisioned for more equitable IWRM. At the same time, it allows for the integration of more mechanistic, process-based ecological information and cultural values in the accounting of value for ecological phenomena. This intuitively appealing concept has great potential for achieving the converging goals of sustainable water resources management and freshwater biodiversity conservation but requires further conceptual development and empirical testing prior to its implementation. We have taken the first steps at establishing the conceptual foundation for its use but acknowledge that empirical testing and further development are crucial to any future contributions by ESA.

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