AQUATIC ECOSYSTEM
HEALTH & MANAGEMENT SOCIETY

MICHIGAN STATE UNIVERSITY PRESS

The urgent need to identify thresholds to use for decisions about shoreline and riparian development in freshwater systems

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Freshwater shorelines, including adjacent riparian habitats, are dynamic intersections between land and water that contribute to the maintenance of biodiversity in both realms. These areas are also affected by multiple stressors at local and global scales, from development to climate impacts. Despite increasing alterations to these areas, often to the detriment of connected ecosystems, and despite many regulations for residential and commercial development, there are no established thresholds across countries and governance levels for how much shoreline or riparian development is too much to maintain freshwater ecosystem function. The urgent need to identify thresholds for shoreline and riparian development in freshwater systems is complicated by a number of challenges, yet there is evidence that threshold effects occur after only a small area of a watershed is developed. Here, we summarize current information on development thresholds for shoreline and riparian areas of freshwater systems. We then discuss the inherent challenges in assigning numeric values to such a diverse set of ecosystems (spanning wetlands, lakes, streams, and more), including considerations such as temporal lags, spatial scales, and cumulative effects. We conclude with a call for research needed to overcome knowledge gaps that will enable practitioners to

Aquatic Ecosystem Health & Management, 27(3): 52–64, 2024.
ISSN: 1463-4988 print / 1539-4077 online. DOI: 10.14321/aehm.027.03.52

apply scientifically-robust thresholds to decisions regarding shoreline and riparian development. Doing so will benefit all actors by providing evidence to support shoreline policies and development guidelines that are inclusive of the aesthetic, recreational, and functional aspects of freshwater systems.

Keywords: cumulative effects, ecosystem, management, regulation, tipping point

Why do we need thresholds for development in shoreline and riparian areas of freshwater systems?

Shorelines are part of the shoreland ecosystem, which includes upland, riparian, and littoral zones (Dennison, 2022). Shorelines and their riparian zones are the interface between distinct but coupled terrestrial and freshwater environments: both provide habitat for a variety of land- and water-based species and they regulate and maintain the physical, chemical and biological conditions of the systems (Schindler and Scheuerell, 2002; Riis et al., 2020; Cooke et al., 2022a), with estimates suggesting at least 70% of vertebrates use riparian habitats at some point in their life (as cited in Naiman et al., 1993). Shoreline riparian areas are also popular for human development due to their aesthetic, recreational, and economic value. While intact riparian zones foster climate resiliency (Cooke et al 2022a), the alteration of near-shore physical habitat often has negative effects on freshwater ecosystems (Lyche Solheim et al., 2013; Teurlincx et al., 2019). For example, riparian alteration affects channel morphology and flow regimes (Del Tánago et al., 2021; Henriques et al., 2022), reduces habitat heterogeneity for terrestrial and aquatic species (Kaufmann et al., 2014; Figure 1), and changes the processes of erosion, filtration, infiltration, noise and light pollution, channel movement, shading, and subsidies (Fisheries and Oceans Canada, 2020). Property owner decisions about altering shorelines are complex (Scyphers et al., 2015), and though education about the benefits of natural shorelines may help reduce site-specific alterations, regulatory restrictions are also important, particularly for managing larger-scale impacts (Norton et al., 2022). Currently, regulators responsible for managing shoreline development have little information available to judge how much development critically impairs ecosystem function. Even so, determining when thresholds for ecosystem health have been reached was identified as the most important topic for freshwater fish habitat management in Canada (Dey et al., 2021). Identifying thresholds will only become more important as effects from climate change become more severe (e.g. Lawrence at al., 2014).

There are many regulations for residential and commercial development, but few for riparian or shoreline alterations across countries and governance levels. In regions with robust governance structure, permits for development are typically issued on a project-by-project basis, but policies that restrict the cumulative number of approved projects in any given area, or guidelines that establish appropriate spatial-temporal scales for consideration, are often lacking. For example, in Canada, cumulative effects were only mandated to be considered in 2019 under the *Fisheries Act*, and so the science to support this policy in practice is still being developed (Department of Fisheries and Oceans (DFO), 2022). Furthermore, in some jurisdictions/countries there are few environmental regulations or, more commonly, enforcement is lacking. We submit that there is dire need to establish thresholds for cumulative shoreline and riparian development in freshwater systems to guide management activities (Jennings et al., 2003; Kelly et al., 2015). First, we summarize examples of development thresholds (i.e. when small changes produce a non-linear – often large – response in an ecosystem component; Samhouri et al., 2010) and how these can lead to ecosystem-level responses. Next, we acknowledge that there are inherent challenges with developing such guidance (e.g. Johnson, 2013; Spake et al., 2022), and finally we discuss the challenges in their application. We conclude with a call for research needed to overcome knowledge gaps that will enable practitioners to apply scientifically-robust thresholds to decisions regarding shoreline and riparian development in freshwater systems. Our team includes researchers with diverse expertise

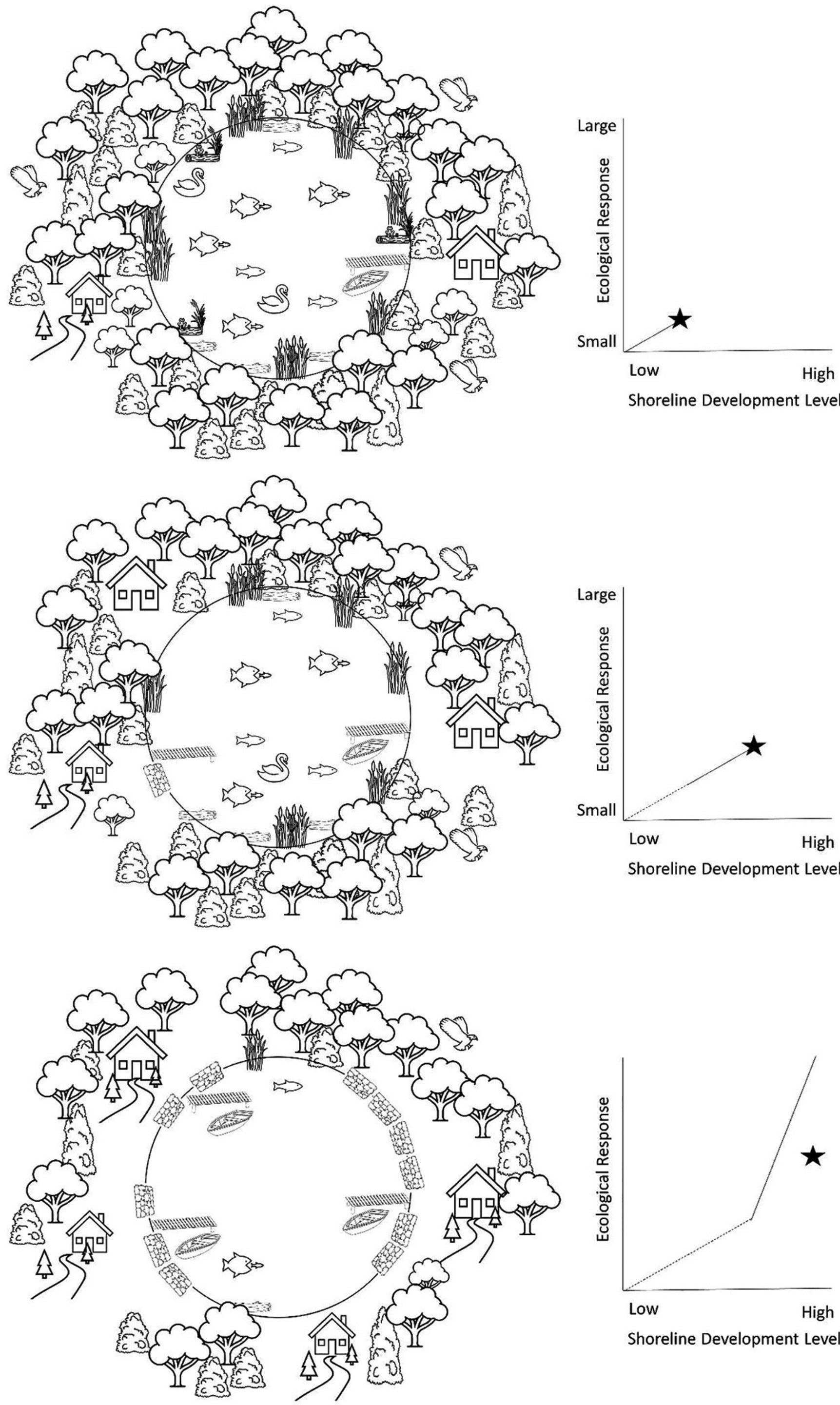

Fig. 1 The relationship between shoreline development and ecological response in a system with a threshold. At low and mid development, (top and middle panel) individual landowner development (represented by the star) reflects the system response (represented by the solid line). As development by individual landowners increases, (bottom panel) there is a point where the linear ecological response hits a threshold and a larger response occurs. Regardless of whether a few landowners do a lot of development or many landowners do small changes, each landowner has abided by regulations, yet the overall cumulative effects are larger than predicted from these individual effects. This may or may not also represent a tipping point in the system.

in freshwater science and management, as well as practitioners and regulators that deal with permit requests. Although we attempt to be global in our thinking, we acknowledge that we are all based in Ontario, Canada and are more familiar with the regulatory framework and management needs in North America. Nonetheless, we are confident that these issues are germane to freshwater systems around the globe.

What is shoreland development?

Here we consider three categories of freshwater shoreland development: 1) shoreline armoring (hard structures preventing erosion)—structures made from sheet piling, concrete, riprap, gabions, boulders, and wood; 2) shoreline alterations (infrastructure in contact with water)—boat ramps, docks, boat houses, and other infrastructure (such as stormwater or tile outlets); and 3) riparian alterations (changes on land)—ranging from cosmetic landscaping such as terrestrial vegetation removal, lawn and garden features, and beach creation, to forestry and agriculture. These categories are often linked as changes in one can create conditions where landowners pursue additional modifications (e.g. removing terrestrial vegetation leads to shoreline armoring). All of these forms of development result in physical changes that alter local chemical and biological conditions including: 1) habitat quantity including connectivity; 2) habitat quality via structure or cover simplification or establishment of aquatic invasive species; 3) water flow and level dynamics (e.g. surface runoff, groundwater connectivity, current diversion); 4) shoreline slope and bank instability and erosion; and 5) nearshore water quality via contaminated run-off (e.g. sediments, nutrients, pesticides) and increased temperature and oxygen demand (reviewed in Fisheries and Oceans Canada, 2020; Brownscombe and Smokorowski, 2021).

For example, shoreline armoring can alter plant, invertebrate, and fish communities and their foodwebs (Doi et al., 2010; Wensink and Tiegs, 2016; Chhor et al., 2020), and armoring generally has a negative effect in soft sediment environments (Dugan et al., 2018). Shoreline alterations can affect vegetation (Sagerman et al., 2020) and fish (Dustin and Vondracek, 2017) communities, and fish behaviour through boat noise (Pieniazek et al., 2020; Fleissner et al., 2022). Alterations to riparian habitat reduces large woody material (LWM) (Pearce et al., 2022) which is also often removed by property owners for aesthetic reasons (Piegay et al., 2005; Le Lay et al., 2008) causing it to be negatively correlated with development at the lake scale (Christensen et al., 1996; Jennings et al., 2003; Wehrly et al., 2012). LWM serves as a refuge, food source, and spawning habitat for fish (Trial et al., 2001; Smokorowski and Pratt, 2007), increases fish community diversity (Talmage et al., 2002), and is also one of the substrates used by periphyton, which are the main food of primary consumers and thus the foodweb (Vander Zanden and Vadeboncoeur, 2020).

In general, shoreland development has many other negative linear effects, a sampling of which include a lower diversity of food items and consumers that reduces trophic links (Rosenberger et al., 2008; Francis and Schindler, 2009; Brauns et al., 2011); fewer frogs (Woodford and Meyer, 2003); less diverse plankton and macroinvertebrate communities (Smith and Kirkwood, 2022); and fewer nesting fish (Reed and Pereira 2009) with altered behaviour (Foster et al., 2016) and community structure (Smokorowski and Pratt, 2007).

What do we know about development thresholds?

We define thresholds as a breakpoint where a larger (non-linear) ecological response occurs in one or several components of the system. In some definitions, a threshold is simply a stopping point along a linear progression of the cumulative impacts of multiple activities (Johnson and Ray, 2021). When single (or multiple) ecological components exceed a particular value (whether linear or non-linear), this may or may not result in a tipping point (where the system moves from one stable state into another) (Kim et al., 2020). Regime shifts and multiple stable states can occur in a wide range of systems (Schroder et al., 2005; but see Hillebrand et al., 2020) resulting in a new system that is often degraded and harder to recover. All these responses can occur simultaneously. For example, riparian development can create tipping points for primary production (e.g. shifts from macrophyte dominance to algal dominance) (Scheffer and Carpenter, 2003), but have a threshold with respect to benthic invertebrates (above which a larger decline happens) (Burdon et al., 2013), and a linear response with respect to amphibians (a constant decline) (Woodford and Meyer, 2003).

There are few examples of non-linear threshold effects related to water quality or habitat conditions in freshwater systems. Forage fish density increased above a threshold of 5 large trees per 30 m shoreline (Brown, 1998). At the watershed scale, >3% impervious (hard surface) cover decreased macroinvertebrate richness in streams (Maloney et al., 2012), and in Canada, it is recommended that watersheds have <10% impervious cover as many components of stream health (fish, plants, amphibians, water quality) become more degraded past that level (Environment Canada, 2013). In wetlands, >10% development affected multiple taxa (Kovalenko et al., 2014), though another study found threshold effects in wetlands were driven by their hydraulic regime (Larsen and Alp, 2015). In small Brazilian streams, the threshold of vegetation loss where fishes and invertebrates were affected varied with stream size (Dala-Corte et al., 2020), and 1-3 kilometers (km) of riparian deforestation (3-20% of watershed area) affected fish assemblages in Appalachian streams (Jones III et al., 1999). In estuarine communities, urban development greater than 3.5-3.7% of the watershed area showed a threshold negative effect on waterbird community integrity (DeLuca et al., 2008), and submerged aquatic vegetation increased over time in sub-estuaries with <5.4% riprap, but not in areas with >5.4% riprap (Patrick et al., 2014).

Costs can also have thresholds. Establishing conservation networks for wetlands prior to development was less costly and resulted in less fragmented networks than trying to establish these areas after natural resource extraction had begun, especially after the developed area reached a threshold of 11% (Cimon-Morin et al., 2016).

On the challenges with identifying thresholds

Challenge 1: Classifying development

Determining how to classify different forms of development is remarkably challenging. For example, in the inland lakes of Ontario, shoreline development is defined as the total number of units (permanent residences, cottages, resorts, trailer parks, campgrounds and camps, and the conversion of forests to agricultural or urban land) within 300 m of the lake or its inflowing stream (Ministry of the Environment, Conservation, and Parks (MOECP), 2019). Some authors use a simple classification by number of houses (defined as buildings with lakefront access or within 10 m of shore) into undeveloped (0 houses km^{-1}), low density (1-10 houses km^{-1}), and high density (>10 houses km^{-1}) categories (e.g. Christensen et al., 1996; Francis and Schindler, 2009; Wehrly et al., 2012). However, does number of units really account for all the changes that development brings? Development can lead to pesticide and herbicide use, impervious surfaces like buildings and new roads, concentrated pet waste, wildlife harassment, septic systems, car tire pollutants, light and noise pollution, boat traffic, human disturbance, etc., that all vary among individual dwellings. Other methods of measuring development include quantifying human activities through energy use (electricity, fuels, fertilizers, pesticides, and water) per unit area per unit time (Brown and Vivas, 2005). Should each individual aspect of development have its own limits, or is housing density a sufficient metric for setting threshold effects of development?

Challenge 2: Scale

Matching the spatial and temporal scale of development with their effects is challenging. Both can be measured at a watershed, lake-wide, or river-reach basis, or a more local scale (e.g. Wehrly et al., 2012) and the future position of the shoreline may change due to wind, waves, and currents (e.g. Tomasicchio et al., 2020). Development can have effects at relatively small linear scales (e.g. 500m: Brauns et al., 2011) and at very large scales (i.e. the Laurentian Great Lakes: Meadows et al., 2005). Moreover, how does scale play out when one considers more mobile organisms (e.g. fish) versus more sedentary organisms (e.g. rooted plants), or ones that have life cycles that require terrestrial habitats across seasons (e.g. turtles, insects)? Does the spacing of development matter? For example, does it matter if the development is all at one end of the lake versus dispersed around the lake? What is the density or spacing of docks that alter fish movement or space use? How do you assess if riparian habitats for egg laying are connected enough to natural shorelines to access those habitats for turtles? Are armoring and alteration equally detrimental, do they occur equally as often, and how often does development or other stressors have synergistic effects (Craig et al., 2017)? What is the lag between a shoreline change now and a community or ecosystem response at some time in the future or at another location? How are continued disturbances considered, such as boat traffic once a dock is in place? Development may have short (e.g. high sedimentation during building phase) and long-term (e.g. changed flow, temperature, nutrient inputs) impacts. As such, how often are both considered? How do we meaningfully include cumulative effects in the face of such lags? These are but a few of the many questions that exist that are relevant to scale.

Challenge 3: Regulations

In North America, housing development is now generally restricted to a minimum 30 m (100 ft) set-back from the high-water mark, but older shoreline developments are situated much closer depending on jurisdiction (e.g. many places have grandfathered regulations that allow boathouse structures at the shoreline: Collison and Gromack 2022) and selective timber harvest is often regulated and allowed (Lee et al. 2004). Activities within that restricted space – such as armoring (hard structures) and alteration (infrastructure and riparian zones) discussed here – may be regulated or simply have best management practices (BMPs), which may vary depending on land ownership (government or private) and level of jurisdiction under consideration. Generally, BMPs recommend a 15-30 m ecological buffer/vegetation protection zone, though this is for mitigating water quality impacts, not for protecting wildlife habitat (Niagara Peninsula Conservation Authority (NPCA), 2022). Riparian zone buffers were originally designed to protect aquatic components by reducing nitrogen runoff from land-use practices (Mayer et al., 2006). Riparian widths to protect aquatic components vary depending on their goal, with 10-30 m a minimum to protect physical and chemical attributes of a stream, 10-50 m for invertebrate diversity, 15-100 m for fish and fish habitat, and 30-100 m for large woody material supply (Broadmeadow and Nisbet 2004; Collison and Gromack 2022). There is less research on buffer width necessary to protect terrestrial components (the native trees and shrubs and their associated mammals, birds, and amphibians) (Lee et al., 2004) and in general buffers needed to protect terrestrial components (100 m) are wider than those for aquatic components (10-30 m) (Wenger 1999). However, the minimal existing science on buffer width effectiveness shows vastly different buffer size depending on focal taxa and impact, ranging from 8 m for water quality protection from herbicides to 1 km for habitat necessary for turtles (reviewed in NPCA, 2022). Though buffers tailor-made for each situation may provide more defensible criteria, this increases complexity, as up to 14 modifying factors have been identified (such as the waterbody slope, size, and type, and presence of fish) (Lee et al., 2004) and so others have suggested fixed-width buffers are clearer and more enforceable (Wenger and Fowler, 2000). Without clear, legally enforceable rules, trained staff to conduct site visits to determine compliance, and centralized documentation of development such as in a registry, effects will continue to accumulate unnoticed.

Binding targets for healthy riparian areas are lacking in many areas. In Canada, a 30 m naturally vegetated riparian area is the government's BMP for streams (Environment Canada, 2013) but there are no guidelines specific to lakes or headwater drainages. The United States' requirement for the assessment of total maximum daily loads (nutrients, sediments, or other impairing factors) for water bodies implies riparian habitats be conserved (Environmental Law Centre, 2021) but buffer width regulation in some US states is still lacking (Mayer et al., 2006). There is also a lack of data for most lakes, especially shallow warm water lakes, and so stream-based guidelines are applied to manage their development. In situations where lakes have a much smaller ratio of land-to-water interface, for example in large lakes or lakes with large contributing watersheds, the scientific defensibility of applying stream-based guidelines becomes a challenge.

How do we balance socioeconomic considerations with ecological thresholds? It may not be possible to apply as strict a threshold as would be ecologically ideal because of competing needs in multi-stakeholder landscapes. For example, in some places, riparian widths are up to the discretion of municipalities where they may be less than 30 m 'for political reasons' (Model Riparian Buffer Protection, 2016). If shorelines are developed, the requirement of offsets to counterbalance the negative effects on fish and fish habitat may be invoked (e.g. in Canada: Government of Canada, 2021). Though this approach may be beneficial, the offsetting measures are determined on a case-by-case basis and have inherent challenges (Coker et al., 2018; Theis et al., 2020; Price et al., 2022; Theis et al., 2022), and riparian connectivity is rarely considered (Environmental Law Centre, 2021).

On the challenges with applying thresholds

Challenge 1: Political and public buy-in

Threshold development points should help guide policy and set standards (Hunter et al., 2009; Kelly et al., 2015), such as supporting the goal of the Canadian federal *Fisheries Act* to offset or counterbalance the harmful alteration, disruption or destruction (HADD) of fish habitat. This would only work in practice if there was a clear scientific basis for the threshold value (or the tipping point, or the maximum allowable cumulative effect) above which no more shoreline development could take place at the individual/lake/reach/watershed scale, so that regulators could say 'no' with more certainty. However, the outcome would be that property owner A can do their development, or that property owner B did a project last year, but owner C who was next in line cannot complete their project because the threshold value has now been crossed and no more development is allowed, a situation that owner C may find hard to understand. Ideally, with solid evidence demonstrating the benefits of such thresholds and clear information campaigns to educate landowners about local limits, greater landowner acceptance would result.

Models such as Ontario's Lakeshore Capacity Model, designed for relating phosphorus loading to shoreline development, could perhaps be modified to include other stressors (Government of Ontario, 2010). These models should also deal with climate change and include new technologies that influence shoreline management (e.g. bubblers in winter that reduce ice cover; light pollution), and this type of flexibility may be especially important if thresholds are mandated by law. Regulators need tools and informed science to consider cumulative impacts at watershed scales (e.g. Meadows et al., 2005; DFO, 2022). In the absence of this, shoreline development and degradation may represent death by a thousand cuts, or the tyranny of small decisions.

Challenge 2: Protection and risk

Protected areas (no development) that include the shoreline can have positive effects on habitat, fish, and songbirds (Nikolaus et al., 2022). Globally, one target is to protect 30% of land and water by 2030 to protect biodiversity and mitigate climate change (Convention on Biological Diversity, 2021). Canadians support higher levels of protection (Wright et al., 2019) and resilient ecosystems provide more benefits especially in the face of climate change (Grantham et al., 2019). However, not all ecosystems are equally at risk, and some are more unique than others (Melles et al., 2014; Hansen et al., 2022). For example, by assessing five major biotic drivers of aquatic ecosystem integrity (energy sources, physical habitat, flow regime, water quality, and biotic interactions) we can use tools (e.g. Ecological Risk Index) to identify the watersheds at highest ecological risk due to human-induced stressors (Mattson and Angermeier, 2007). Globally, there are calls to create a prioritization framework to identify degraded ecosystems (e.g. the IUCN Red List of Ecosystems: IUCN-CEM, 2022), as by 2030 the Convention on Biological Diversity (2021) has the goal to restore 20% of degraded ecosystems with a focus on priority ecosystems.

Challenge 3: Landowner education and buy-in

Is the promotion of sound stewardship practices through landowner education and grant and tax incentives more effective than regulatory restrictions? Property owners often want things that are not compatible with a healthy watershed, and misperceptions are common (Scyphers et al., 2015). For example, natural or unaltered shorelines are paradoxically perceived by the public as being less durable and requiring more maintenance than vertical walls (Scyphers et al., 2015), and shoreline property owners often want one consistent water level. How do we convince property owners that water level fluctuations are important for healthy habitats? How do the actions of one property owner affect others? For example, homeowners may alter their shorelines in response to their neighbor's shoreline activities as a form of social conformity (Goddard et al., 2013). However, shoreline alterations can also be made in response to scouring or erosion caused by a neighbor's shoreline alterations (Scyphers et al., 2015). These issues can be perpetuated by contractors speaking to neighboring owners about the perceived benefits of armored solutions and the cost-saving if they do the work now while they are in the area. This can also be exacerbated by a lack of contractors trained in bioengineering solutions aimed at reducing ecological impacts. Eco-engineering solutions and living shorelines are being used more and more, though are currently focused mainly in coastal marine systems (e.g. Morris et al., 2018; Smith et al., 2020). However, the very nature of natural coastal variability that generates resiliency in these systems is often seen as a detriment compared to engineered structures that result in constancy and predictability but no other ecosystem services; the burden is then placed on those proposing these more natural approaches to show they actually work and are cost-effective. Still, the promotion of these 'greener' solutions may increase the desire for development by homeowners or its acceptance by regulators, possibly spurred by the assumption that better engineering solutions are a panacea for whatever is altered by development.

A call to action for identifying and applying thresholds in practice

The idea of ecological thresholds is appealing, as it implies there is a tangible, transparent, objective, consistent, and non-controversial decision-making process (Johnson and Ray, 2021). Yet there is a lack of general principles on how to determine which variables are reliable, measurable, and responsive on the appropriate timescales, and so would be appropriate as indicators across a wide range of systems. In some cases, indicator or umbrella species may be used as sentinels for threshold effects. Appropriate variables are crucial if we are to monitor systems to know when thresholds are being approached (Kelly et al., 2015), especially considering thresholds are context dependent (e.g. naturally oligotrophic vs. eutrophic systems) and have so far mainly been established in specific systems. We echo Spake et al.'s (2022) call for researchers to identify early-warning signals (e.g. increases in variance), for policy makers to be proactive instead of reactive (responding to early warnings instead of waiting for degradation), and for everyone to identify the underlying processes and drivers at relevant temporal and spatial scales to help create scientifically based guidance and tools that can be used by managers and regulators (e.g. Stutter et al. 2021). We suggest that greater collaboration of experts across disciplines (natural sciences, social sciences, policy, etc.), knowledge systems, organizations, and regions is key to incorporate the best available information and address deficiencies identified here (e.g. Cooke et al., 2022b).

Adaptive management will be key in determining the sensitivity and likelihood for any particular system to be in danger of approaching a tipping point, and socioeconomic interests will play a role in determining acceptable risk (Johnson and Ray, 2021). Once threshold values are established, then complementary restoration targets would help rectify systems that have fallen below the threshold, though multiple stressors (which are the norm in freshwater systems; Reid et al., 2019; Spears et al., 2021) may have to be alleviated for restoration to be effective (Allan et al., 2013) and this may take centuries (Moreno-Mateos et al., 2020). Shoreline restoration incentive programs exist, such as revegetating, not mowing, or bioengineering where there are valid erosion issues, and so damaged areas can be restored, but simply restoring benchmark physical habitat or water quality parameters may fail to ensure ecological processes such as foodweb interactions are also restored (Albertson et al., 2018). As a society, we should embrace the response hierarchy of avoid > reduce > reverse when considering ecosystem degradation (Cowie et al., 2018).

Conclusions

The loss of biotic integrity compared to reference conditions in human-altered freshwater ecosystems is ongoing. Practitioners and regulators play an important role in permitting activities and we all share in the desire to maintain functioning freshwater ecosystems (Twardek et al., 2021). Some patterns are already emerging, such as the large riparian buffer width necessary to protect both aquatic (30-100 m) and terrestrial (up to 1 km) components (NPCA, 2022), and the evidence for adverse threshold effects occurring when just 3% of land in a watershed is developed (DeLuca et al., 2008; Maloney et al., 2012). By identifying best practices for our shorelines and identifying thresholds beyond which changes are disproportionately large, we can balance human desires with ecosystem function, which benefits everyone and everything in the end.

Statements and declarations

Funding was provided by the Natural Sciences and Engineering Research Council (Alliance Missions Grant led by SJC), the Rideau Valley Conservation Authority, Kawartha Conservation, and Fisheries and Oceans Canada. The authors have no relevant financial or non-financial interests to disclose. We are grateful to several anonymous referees for providing thoughtful comments on our manuscript.

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