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CONCEPTS & THEORY

A strategy for optimizing catchment management actions to stressor–response relationships in freshwaters

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Abstract. A myriad of management actions can be applied to reduce anthropogenic pressures on aquatic environments. Appropriate management actions, whether they be mitigations of contaminant transfer to receiving environments or interventions within the receiving environments to alter resilience to a contaminant, are those which are acceptable to stakeholders and cost-effective and which operate over desired time frames. The stressor–response relationship describes the change in ecological, social, or economic value of a receiving environment when impacted by a specific contaminant. Defining a receiving environment × value × contaminant system and determining a specific stressor–response relationship for that system provide valuable decision support strategy to optimize management actions toward a water quality objective. Here, we outline a potential method for using stressor–response relationships to help identify the most appropriate management actions for aquatic ecosystems. We use the example of a eutrophic lake to show how the method can be applied to any receiving environment × value × contaminant system.

Key words: *Escherichia coli*; intervention; mitigation; nitrogen; phosphorus; sediment.

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Introduction

Agricultural production, and particularly livestock farming, has been linked to the deterioration of soil and water quality (Steinfield et al. 2006). Pressures on soil and water can be measured directly (or via proxies) in the form of concentrations or loads of contaminants such as the fecal indicator bacterium *Escherichia coli*, nitrogen (N), phosphorus (P), and eroded sediment (Muirhead and Monaghan 2012, Basher 2013, Larned et al. 2016). These contaminants come from a wide variety of sources including fertilizers (Ledgard et al. 1999), tillage (Withers et al. 2007), and livestock (Holz 2010). Contaminants are linked to receiving environments by pathways that include direct deposition (Miller et al. 2014), eolian erosion and transport (Li et al. 2004), surface runoff and shallow subsurface flow (including interflow, preferential flow, and artificial drainage; Heeren et al. 2010, Mellander et al. 2016, Monaghan et al. 2016), and groundwater flow (Holman et al. 2010, Scarsbrook and Melland 2015).

A large number of management actions are in use, alone and in combination, to reduce the loss of contaminants from land or lessen their impact in receiving environments. In this paper, we use the term mitigations to refer to management actions that reduce the loss and transfer of contaminants from primary enterprises (i.e., farms, production forests, orchards, rangelands) to receiving environments (i.e., lakes, estuaries, rivers, aquifers), and the term interventions to refer to management actions that increase the resilience (i.e., lessen the impact of contaminants) of receiving environments to contaminant loads. We use the term management actions to refer to mitigations and interventions collectively.

Studies of management actions commonly assess effectiveness in terms of decreases in contaminant load and recently have begun to focus more on the financial costs or the time required for a given management action to have maximum effect (treatment speed; Bailey et al. 2013, Gooday et al. 2014, Schoumans et al. 2014). However, costs vary from one jurisdiction to another due to a range of factors including the availability of labor, the cost of raw materials, field or catchment conditions, soil types, and policies such as incentive schemes or subsidies (Roberts et al. 2012, Vinten et al. 2017). Furthermore, although proxies such as connectivity indices exist to indicate the likely response time of actions, direct evidence of treatment speeds is sparse (Stieglitz et al. 2003, Leibowitz et al. 2018). As a result, there is a moderate level of uncertainty among land managers, investors, and regulators about the relative performance of different on-farm mitigations and within-receiving-environment interventions that may be included in farm and catchment plans and implemented to meet water quality objectives (Payne and White 2006). Failure to advise on the performance (cost, effectiveness, and other factors) of management actions may lead to some, for example—farmers, underestimating the adverse effects of their farming activities on receiving environments, or paying or doing more than necessary to achieve water quality objectives in receiving environments (Daigneault et al. 2017).

Among rural land uses, livestock farming has the greatest range of contaminants and contaminant loss pathways due to the grazing of animals (e.g., discharging microbial pathogens to streams via excreta) and the wide range of soils and climates used for livestock farming (Steinfield et al. 2006). Among livestock systems, farms that graze livestock outside year-round also require a greater diversity and range of management actions to reduce contaminant losses from pathways active in winter and/or summer compared to confined animal feeding operations that can store manure and apply it when and where runoff is unlikely or hybrid farms that only graze for a portion of the year (Gourley and Weaver 2012, Kirkegaard et al. 2014). Hence, an objective approach is needed to provide advice on matching management actions both to farm production systems and to the assimilative capacities of different receiving environments, especially in grazed livestock systems. Here, we define assimilative capacity as the maximum contaminant load that can be discharged into a receiving environment while achieving a water quality objective (Cairns 1998). Some farmer- (Payne and White 2006), industry- (DairyNZ 2015, Dairy Australia 2016), and government-led (Ministry for the Environment and Ministry for Primary Industries 2015) initiatives give advice on cost and effectiveness of mitigation actions. However, none of these initiatives account for the assimilative capacities of different receiving environments, and none consider the benefits of interventions in receiving environments as alternatives to or in addition to mitigation actions. This paucity of information about the interaction between mitigations and interventions is a major gap in the literature and could result in overly restrictive regulation on land use practices to achieve good water quality outcomes in receiving environments where there exists either a large assimilative capacity or scope to implement cost-effective interventions.

In this conceptual paper, we (1) summarize and assign scores to the costs, effectiveness, and treatment speeds of each of a range of management actions available for use in countries with large areas of grazed livestock farming, using Australia and New Zealand as examples, (2) outline some of the factors important in minimizing the costs of these management actions and maximizing their effectiveness and treatment speeds, (3) propose a process for assessing environmental impacts of contaminants in different classes of receiving environments based on stressor–response relationships and for scoring the value of a given response variable, the state of a given stressor, and the slope of the corresponding stressor–response curve, and (4) propose a strategy for selecting effective management actions based on the assessment of environmental impact in the preceding step, and whether the objective for a receiving environment is to avoid degradation or initiate recovery. This strategy is intended to help land managers, investors, and regulators to make informed decisions such as applying mitigations to prevent contaminant loads from exceeding load limits, or proceeding with land use developments that are acceptable because current contaminant loads are below the load limit.

Management Actions Available for Grazed Livestock Farming

Information was compiled on mitigations and interventions available in Australia and New Zealand for grazed livestock farming. Although there are many potential water quality contaminants that originate from land-based activities (e.g., heavy metals, pesticides), our focus was on the four contaminants highlighted most often in policy (Ministry for the Environment 2014) and by the public (Hughey et al. 2013): nitrogen, phosphorus, sediment, and *E. coli*.

In order to provide a consistent approach across all management actions (Appendix S1: Tables S1 and S2), we only included actions that met the following criteria:

1. Data used to define the mean cost and the range of effectiveness for each action were available for three or more observations;
2. The mode and speed of treatment were known;
3. The action was applicable over a wide geographic range; and
4. Where there was high variability in cost, effectiveness, or speed of treatment, the reasons for the variability could be accounted for when assigning the applicability of measures to different farming systems and receiving environments.

We evaluated and scored management actions based on their cost and effectiveness to reduce each contaminant. Effectiveness was expressed as a percentage reduction in contaminant loads retained or reduced through mitigations or immobilized by interventions. Cost was expressed in $ per ha for the target contaminant, but normalized to a loss of 1 kg of nitrogen, phosphorus, or sediment, and 10^{12} coliform-forming units for *E. coli*. For the purpose of demonstrating the strategy, cost and effectiveness of management actions were grouped into low, medium, or high categories by dividing cost and effectiveness into thirds and assigning their mean value scores of 1–3, respectively. For reference, the range of cost for each mitigation is given in Appendix S1: Table S1. Management actions were also categorized and assigned scores according to their likely treatment speed as slow (1), moderate (2), and fast (3), which correspond to time periods required to achieve maximum effect of >1 yr, <1 yr, and <1 season, respectively. These treatment speed categories reflect actions that could be incorporated as part of tactical (day-to-day) or strategic (seasonal to yearly) management decisions. To implement the strategy, we derive composite scores as the product of the individual cost, effectiveness, and treatment speed scores. This yields a wider range of possible composite scores than summing the scores. However, we recognize that the scoring system may change after testing, when new actions are developed, or as costs change. We also recognize that the implementation of management actions is context-specific and hence recommend that the array of suitable actions be confirmed or re-assessed for each application of the strategy set out below.

An example assessment of cost and effectiveness is presented for phosphorus mitigations in Fig. 1, and for nitrogen, sediment, and *E. coli* in Appendix S2: Fig. S1. For clarity, both figures show mitigations, but exclude interventions. Additional information on mitigations and interventions, including their treatment speeds, is given in Appendix S1: Tables S1 and S2.

We only estimated the cost, effectiveness, and speed of treatment for interventions that were directly related to contaminant control (e.g., removal of bioavailable nitrogen through denitrification, long-term sequestration of phosphorous in sediments). However, we recognize that other interventions have indirect effects on contaminants by reducing contaminant impact. Examples of indirect interventions include food-web biomanipulation (Burns et al. 2014) and removal of algal proliferations using algicides, sonication, and flushing flows (Rajasekhar et al. 2012).

Costs for mitigations and interventions for rivers and streams were expressed in dollars per unit of contaminant contributing area per year

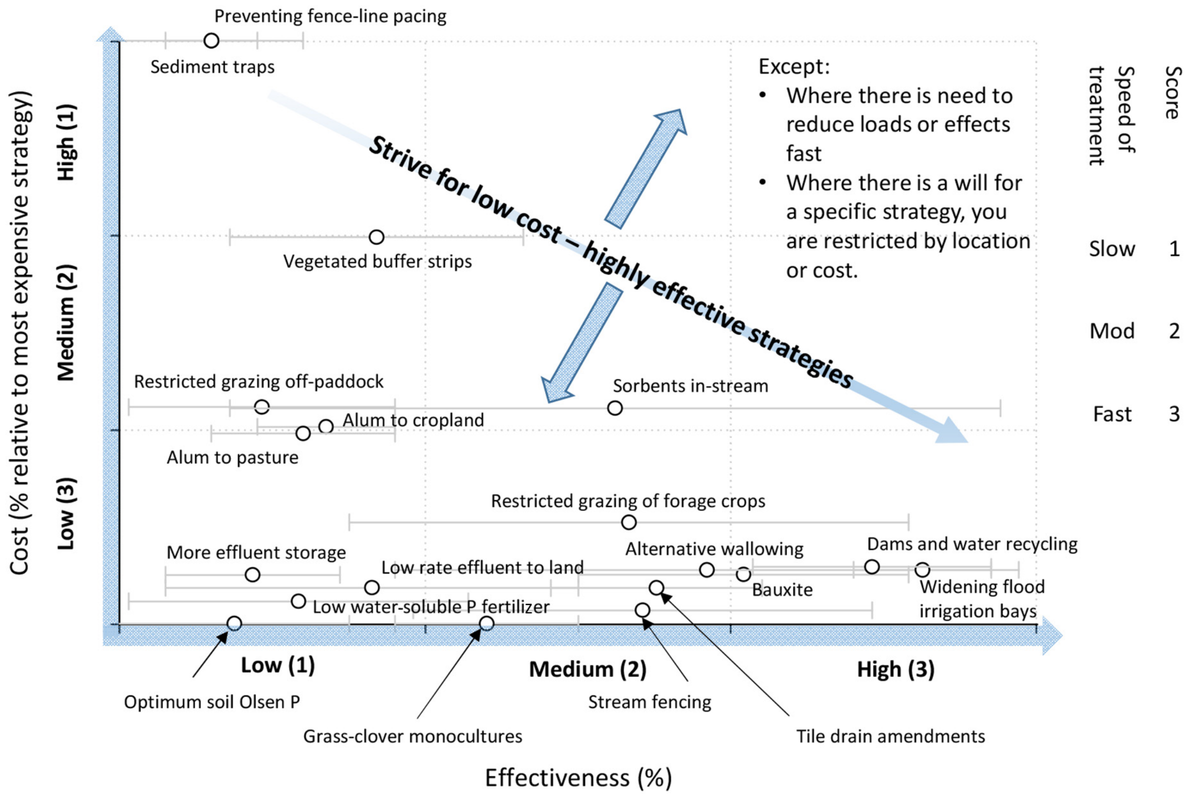

Fig. 1. The relationship between mean cost and effectiveness of phosphorus mitigation and management actions along with the likely speed of treatment. A range is given for effectiveness which is largely due to variation in edaphic conditions, but not for cost where variation is likely to be greater due to factors in addition to edaphic conditions such as the cost of labor. Composite scores for management actions are calculated as the product of scores for cost, effectiveness, and speed of treatment classes.

($/ha of contributing catchment/yr). Because costs for lake interventions can occur independent of the contributing catchment area, costs were expressed in $/ha of treated lake/yr.

Factors to Consider When Choosing Management Actions

Land managers and investors aim for low-cost, highly effective, and fast management actions to minimize their environmental footprint and maintain a social license to operate within a regulatory framework. Using the scoring system set out above, this aim can be numerically expressed as a composite score: low cost (3) × highly efficient (3) × fast treatment (3) = 27. Graphically, this aim corresponds to the lower right corner of Fig. 1. However, there are at least four exceptions to the assumed aim, when

1. There is a need to reduce contaminant loads regardless of the cost;
2. There are large maintenance costs for a given management action;
3. Suitable sites for management actions are restricted (e.g., by covenants); and
4. There is pressure to use a particular mitigation or intervention action.

The first two exceptions would encourage the use of actions located toward the top right of Fig. 1, while exceptions 3 and 4 would encourage the use of actions toward the bottom left of Fig. 1. Beyond these exceptions, there are a number of other issues relating to cost, effectiveness, and treatment speed that should be considered when selecting and optimizing management actions. These considerations are explained below.

Applicability

Management actions may be applicable to only one class of enterprise or receiving environment. For example, in Australasian grazed livestock systems, the use of low-rate dairy shed effluent application to land to mitigate contaminant losses is generally restricted to dairy farms (Houlbrooke et al. 2004).

Placement and timing

Targeting mitigation actions to localized critical source areas, which account for the majority of contaminant losses, greatly increases their cost-effectiveness (Doody et al. 2012, Vinten et al. 2017). McDowell (2014) showed that across 14 sub-catchments farmed with sheep, red deer, beef, or dairy cattle, the cost-effectiveness of mitigations targeted to critical source areas was six to seven times higher than untargeted mitigation actions. The effectiveness and placement of mitigations can also be influenced by the interaction of multiple contaminant flow paths through a catchment (Cumming 2011). For example, knowledge of denitrification zones could be exploited to plan where mitigations (e.g., cover crops, crop rotations, establishment of wetlands) would be best placed to reach a 20–38% reduction in nitrogen load with much greater certainty than applying actions across the whole catchment an untargeted manner (Hashemi et al. 2018*a*, *b*).

Co-benefits

When implementing management actions, there may be a tendency to focus on a single contaminant, but this neglects the possibility that some management actions reduce the loss, transport or impact of multiple contaminants (i.e., co-beneficial management actions). For example, many of the mitigations that target phosphorus and sediment losses can also decrease pathogen loads. Richkus et al. (2016) estimated that actions put in place to meet the total maximum daily load in the Chesapeake Bay for phosphorus and sediment could decrease pathogen load by 19–27%. Gasper et al. (2012) estimated that these actions could also play a significant role in sequestering greenhouse gasses. McDowell et al. (2017) compared two approaches to mitigating N and P losses from grazed dairy farms in four regions of New Zealand: one to target a single contaminant with the most cost-effective measure and a generalized approach that considered costs, ease of implementation, and their effectiveness for multiple contaminants. The targeted approach decreased losses of the target contaminant faster than the generalized approach, but other contaminants were not reduced. Targeting single contaminants could jeopardize water quality and ecosystem health should other contaminants become important downstream (McArthur et al. 2010) or over time.

Longevity

It is often desirable to employ management actions that remain effective over long time frames. Furthermore, the effectiveness of some actions decreases with time, especially if the baseline changes from the legacy of past land use resulting in a time lag caused by the slow movement of a contaminant through the catchment (Meals et al. 2010, Meter and Basu 2017), or the stressor–response relationship changes. For example, the efficacy of wetlands to remove phosphorus from flowing waters decreases as wetlands fill with sediment, or anaerobic conditions develop that remove N via denitrification desorb sediment-bound phosphorus into the water column (Ballantine and Tanner 2010). Changes may also occur due to climatic variability, that may induce more frequent and intense storms that deliver greater contaminant yields (Ockenden et al. 2016, 2017). Depending on the magnitude of change, the cost-effectiveness of actions may be compromised. Indicators, such as a significant change in the load of contaminant lost from different land uses should be used as a trigger to determine whether additional trials are required to confirm the performance of actions under a changed climate.

Uncertainties in cost-effectiveness

A poor understanding of contaminant sources and pathways and impacts of contaminants on receiving environments increases uncertainty in estimates of cost-effectiveness of management actions, which in turn increases uncertainty when recommending management actions (Wainger et al. 2013). To accommodate some uncertainty, but still allow decisions to be made, management actions should only be implemented when there is prior evidence of cost-effectiveness under similar conditions. Further accommodation is made by reporting the costs and effectiveness of management actions as ranges (Gooday et al. 2014, Geng et al. 2015, Daigneault et al. 2017), which recognizes that despite having similar conditions, the performance of a given management action will vary.

Unintended consequences

The unintended consequences associated with applying a management action can be significant. For instance, the nitrification inhibitor dicyandiamide (DCD) was used on grazed and fertilized pastures to mitigate the leaching of nitrogen from New Zealand soils. However, DCD can leach from soils to groundwater and emerge in surface water, where suppression of nitrification can lead to enriched ammonia concentrations (Smith and Schallenberg 2013). In meeting the bioenergy goals of the United States, Adusumilli et al. (2014) estimated that significant increases in erosion and nutrient loading were likely if pastureland was converted to intensive biomass production for biofuel. To minimize unintended consequences, it is important to increase our knowledge of the potential negative side effects of management actions.

The Missing Factor: Accounting for Stressor–Response Relationships in Receiving Environments

Simple stressor–response relationships for receiving environments can be linear or non-linear, potentially showing a wide range of sensitivities or slopes. The state of a stressor variable (along the x-axis) and the value of a response (along the y-axis) can both be divided into three classes. An example of a scoring of a stressor–response relationship for a class of receiving environment is shown in Fig. 2. Each value, stressor state, and sensitivity (slope in the direction of degradation or recovery) is assigned a score from 1 to 3.

The scoring scale depends on whether the objective is to (1) avoid or minimize the risk of degradation in the value of a response variable or (2) to initiate recovery after degradation has occurred. If the objective is to avoid degradation, a response variable most at risk of degradation would have the greatest score (value score = 3), the stressor would have the greatest score (state score = 3), and the stressor–response relationship would indicate high sensitivity (i.e., a steep slope; sensitivity score = 3), resulting in a composite score of

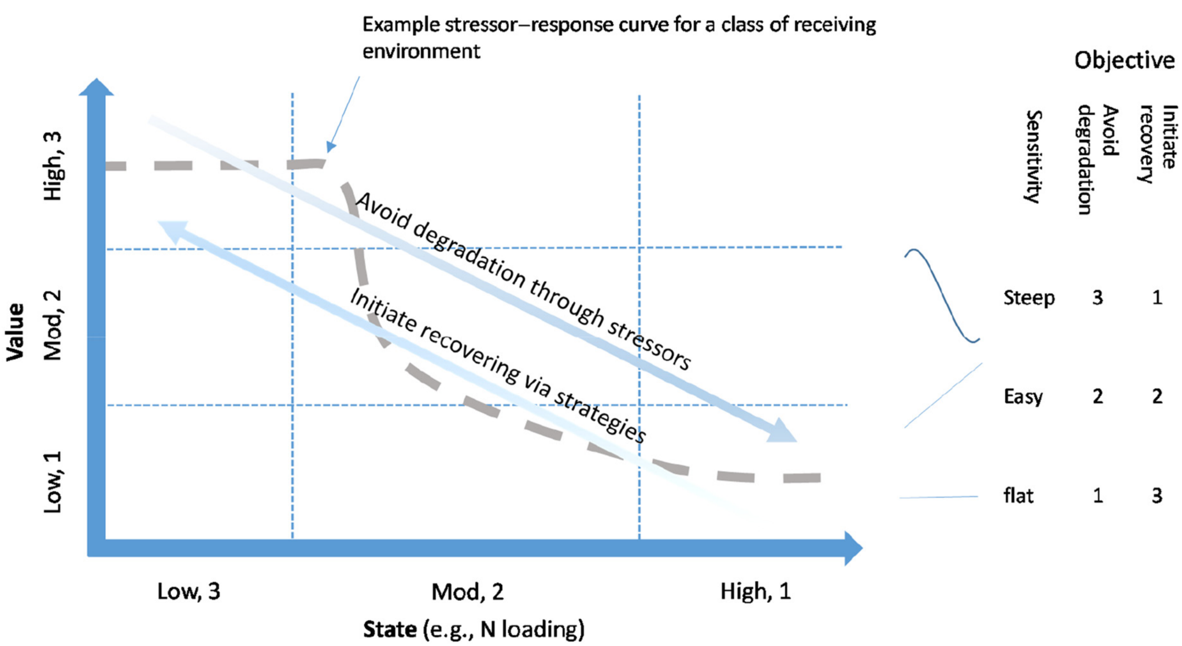

Fig. 2. Conceptual diagram showing the shape and scoring (1–3) of the value, state, and sensitivity of a stressor–response relationship for a given receiving environment. Scoring for sensitivity is dependent on whether the aim is to avoid degradation or initiate recovery.

$3 \times 3 \times 3 = 27$. In contrast, a response variable with the best prospect for recovery would have a low score (value score = 1), the stressor would have the lowest score (state score = 1), and the stressor–response relationship would indicate high sensitivity (sensitivity score = 1), resulting in a composite score of $1 \times 1 \times 1 = 1$. Note that the scale for scoring the sensitivity of the stressor–response curve is inverted depending on the objective (avoid degradation or initiate recovery).

At this stage, there are two composite scores—one expressing the cost-effectiveness and treatment speed of management actions (Fig. 1), and another expressing value, state, and sensitivity as a position on a stressor–response curve for a receiving environment (Fig. 2). These two composite scores can be used to calculate a benefit quotient that has the composite score for management actions as the numerator and the composite score for the receiving environment as the denominator. The benefit quotient in turn expresses the prospect for achieving a management objective based on a given stressor–response relationship for a given receiving environment. Thus, the range of potential values for the benefit quotient is 1/27 to 27. For management actions, a composite score close to 27 (e.g., low cost, highly efficient, and fast treatment) is desired. For a receiving environment, the composite score to use in the benefit quotient will depend upon whether the objective is to avoid (or minimize) the risk of degradation in the value of a response variable or to initiate recovery after degradation has occurred. To minimize the risk of degradation, the

Fig. 3. Photograph showing the extent of a *Ceratium* sp. bloom on Lake Hayes, Otago, New Zealand, during the summer of 2015/2016. Photograph used with the permission of Fish and Game, New Zealand.

best composite score would be 27. Thus, a benefit quotient of 1 would indicate a good prospect of the management action achieving the objective of avoiding degradation. In contrast, to initiate recovery, the best composite score would be 1. Thus, a benefit quotient of 27 would indicate a good prospect of the management action resulting in a recovery from a degraded state.

Application of the Management Strategy

The application of the management strategy detailed above is summarized in the following steps:

1. A water quality objective is set to avoid degradation or initiate recovery in the value of a response variable relative to a single contaminant or group of contaminants;
2. The appropriate stressor–response curve for the receiving environment and contaminant (or group of contaminants) in question is identified and the current position on the curve (Fig. 2) assigned a composite score is calculated as the product of the value, state, and sensitivity of the system;
3. Management actions are assigned a composite score for product of their cost, effectiveness, and treatment speed and their applicability for local conditions assessed;
4. The benefit quotient is assessed as the quotient of the composite score of actions over the composite score for the receiving environment and each action ranked against a stakeholder's ability to implement actions after accounting for factors outlined in section 2 such as timing, placement, longevity, co-benefits, and unintended consequences. Advice is also given on how to monitor the performance of the management actions in terms of achieving their objective.

As an example, we show how the strategy could be applied to Lake Hayes, Otago, New Zealand. The lake (surface area = 270 ha, maximum depth 31 m) lies in a small catchment dominated by sheep and red deer farming (Caruso 2001). Beginning in the late 1960s, the lake has suffered regular, severe, summer phytoplankton blooms, due to phosphorus enrichment (Fig. 3), but these have not developed in two of the last seven summers. It

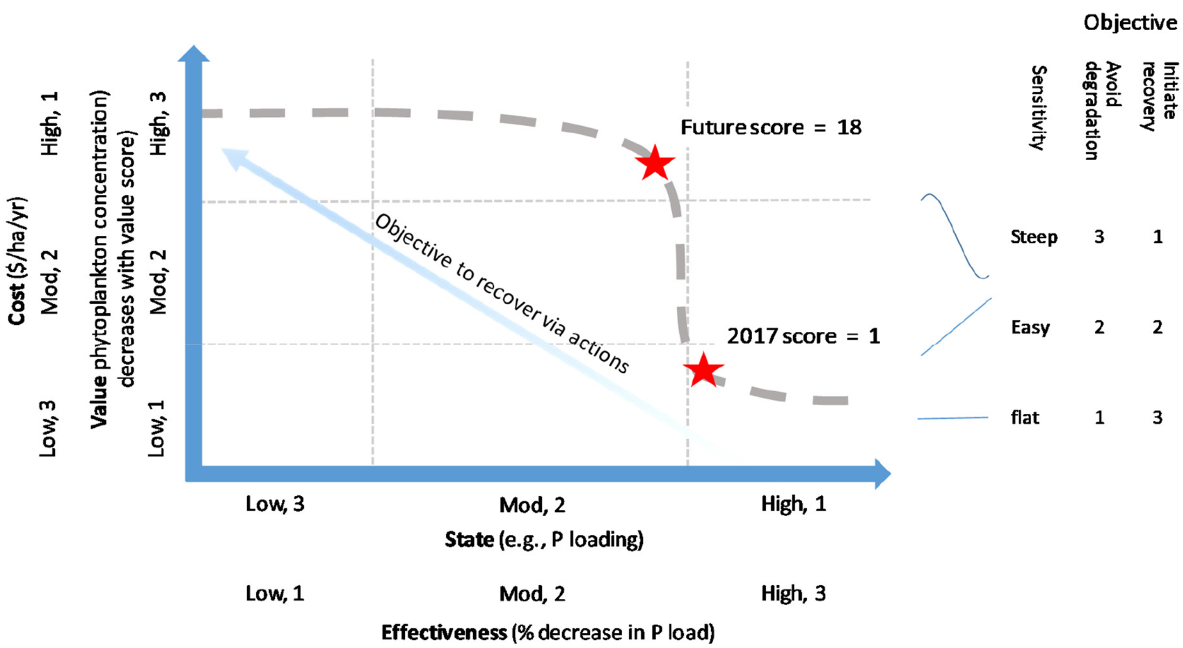

Fig. 4. Conceptual diagram showing the overlap shape of cost and effectiveness (speed of treatment not shown) of management actions and the stressor–response curve for phosphorus and phytoplankton in Lake Hayes (Otago, New Zealand) showing the scoring of value, state, and sensitivity of the lake in 2017 and at a point in the future where progress toward the objective of recovering the lake is being achieved.

has been suggested that the lake is approaching a nutrient recovery threshold characterized by a logistic relationship (Fig. 4) between phytoplankton biomass and nutrient loading (Schallenberg and Schallenberg 2017). If no relationship was available for the lake, it would be reasonable to forecast a likely trajectory or use a relationship for a similar type of receiving environment. Forecasting the trajectories of relationships is a rapidly growing area of research (Flynn et al. 2015, Petchey et al. 2015, Dietze et al. 2018).

In step 1, we assessed that the lake was a good candidate for treatment given its current position on the stressor–response curve and define a hypothetical objective to ensure recovery by reducing phosphorus loading and phytoplankton growth. In step 2, the composite score (1) for the receiving environment was calculated as the product of a lake in poor state (score = 1), of low value (score = 1), but sensitive to small changes in phosphorus loading (score = 1). In step 3, composite scores were given for the cost, effectiveness, and speed of treatment of potential management actions identified according to their applicability for local conditions (3–27; Table 1). The final step saw benefit quotients generated for the use of mitigation and intervention actions in the lake that ranged from 3 to 18 and 6 to 18, respectively. Before implementing actions, additional information would be given to stakeholders on where to monitor P loads in the catchment and phytoplankton in the lake to ensure that the management actions were working, and on how those actions would be influenced by factors in section 2 such as longevity (including climate change) and uncertainties. The benefit quotients and additional information inform decisions on which management actions will achieve the best cost, effectiveness,

Table 1. Name, description, score, and benefit quotient of suitable interventions and mitigation actions (from Appendix S1: Tables S1 and S2) in recovering Lake Hayes in 2017 and, in parentheses, at a point in the future.

Suitable action	Description	Action score	Initial and (future) benefit quotient
Mitigations			
Stream fencing	Preventing livestock access to stream decreases stream bank damage (and sediment inputs via bank erosion) bed disturbance of sediments (and entrained *E. coli*, N, and P) and stops the direct deposition of excreta into streams	27	27 (1.5)
Restricted grazing of forage crops	Restricted grazing of a forage crop in winter to reduce deposition of excreta and surface erosion by grazing animals	18	18 (1)
Alum to pasture	P-sorbing aluminum sulfate (alum) sprayed onto pasture a week before grazing to prevent subsequent surface runoff losses of P	12	12 (0.67)
Alum to grazed forage crops	P-sorbing aluminum sulfate (alum) sprayed onto a winter forage crop just after grazing to prevent surface runoff losses of P	12	12 (0.67)
Optimum soil test phosphorus concentration	Matching soil Olsen P concentrations to pasture and forage crop requirements avoids enriched soil P concentrations that are more likely to lose more P in runoff compared to that at an agronomic optimum concentration	3–9	3–9 (0.16–0.5)
Preventing fence-line pacing	Tree planting to provide shelter and maintaining sufficient feed to avoid stress when, for example, when feed is low or near calving	6	6 (0.33)
Vegetated buffer strips	Vegetated buffer strips work to decrease contaminant loss in surface runoff by a combination of filtration, deposition, and improving infiltration	6	6 (0.33)
Sediment traps	In-stream traps allow coarse-sized sediment and associated N and P to settle out	3	3 (0.16)
Interventions			
Phosphorus inactivation or flocculation	Chemicals such as alum (aluminum sulfate) can "lock up" dissolved phosphorus in lakes via adsorption and precipitation processes	18	18 (1)
Lake hydraulic flushing	Reduce natural residence time, reduce internal loading, and increase throughput of nutrients and sediment	6–18	6–18 (0.33–1)
Aeration, oxygenation, and destratification	Air/O_2 pumped to the bottom of lakes to destratify lakes and prevent P release under reducing conditions	9	9 (0.5)

and treatment speed according to the stressor–response curve's indication of state, value, and sensitivity of the lake. If the objective had been to avoid degradation, with the lake being scored in the top left sector of Fig. 4, the composite score would have been 9 (low phytoplankton concentration [3], low P loading [3], and flat sensitivity [1]). Benefit quotients would range from 0.33 to 3, but the order of actions would be the same as per the objective of initiating recovery. However, let us also consider a case in the future where the lake is scored along the same stressor–response curve as having high value (score = 3), acceptable state (score = 2), but still highly sensitive (score = 3). Under the objective of initiating recovery, the composite score for the lake would be 18. Applicable management actions would yield benefit quotients for mitigation and intervention ranging from 0.06 to 1 and 0.33 to 1, respectively. These benefit quotients indicate that there is less incentive for pursuing actions for further recovery, but highlight those actions that should be maintained should the objective shift to avoiding degradation.

Conclusions

We provide a simple method to score the costs, effectiveness, and treatment speeds of management actions used to mitigate the loss of water quality contaminants from land or to lessen their impacts in receiving environments. Following the identification of a stressor–response curve for a receiving environment, a similar method is used to assign a composite score to the value of the response variable, the state of the stressor, and the sensitivity of the stressor–response relationship. Once a water quality objective has been set to initiate recovery or avoid degradation of a response variable, dividing the composite scores for potential management actions by the composite score for the receiving environment (based on its stressor–response curve) yields a benefit quotient that identifies management actions that can best achieve the objective. Along with additional information about the suitability and longevity of management actions, the benefit quotient can be used to prioritize management actions, enhance the creation and implementation of catchment plans, and along with monitoring of response variables—gauge progress to improve water quality while maintaining primary productivity.

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