Journal of Environmental Management 233 (2019) 218–237

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

A mixed-methods approach to strategic planning for multi-benefit regional water infrastructure

Sasha Harris-Lovett[a,c,*], Judit Lienert[b], David Sedlak[c,d]

[a] Energy and Resources Group, University of California, Berkeley, United States
[b] Eawag: Swiss Federal Institute of Aquatic Science and Technology, Switzerland
[c] Engineering Research Center for Reinventing the Nation's Urban Water Infrastructure (ReNUWIt), United States
[d] Department of Civil and Environmental Engineering, University of California, Berkeley, United States

ARTICLE INFO

Keywords:
Multi-criteria decision analysis
Regional environmental planning
Nutrient management
Wastewater treatment
Stakeholder analysis

ABSTRACT

Finding regional solutions for water infrastructure and other environmental management challenges requires coordination, communication, and a shared understanding among different stakeholders. To develop a more versatile and collaborative decision-making process for nutrient management in the San Francisco Bay Area, we used a mixed-methods approach consisting of stakeholder analysis with cluster analysis, multi-criteria decision analysis (MCDA), and scenario planning. These methods allowed us to identify agreements and disagreements in stakeholder objectives and preferences, clarify ways in which different options could meet the goals of diverse stakeholders, and elucidate how scientific uncertainty about technical performance and future conditions could affect management strategies. Results of the analysis indicate that several non-conventional nutrient management options like constructed wetlands and increased water recycling for irrigation met the goals of many stakeholders under a variety of future scenarios. A comparison of MCDA results with a more traditional 'cost-efficiency' measure (i.e., optimizing for the lowest cost per mass of nutrients removed) revealed little correlation between the two methods for stakeholders who expressed a preference for co-benefits of management options such as increased water supply and nutrient recovery for fertilizer use. The method also allowed us to identify key areas of disagreement (e.g., the relative importance of constructing infrastructure that would not be affected by sea level rise) that should find regulatory or professional consensus before advancing with decision-making. This mixed-methods approach is time-consuming and requires specific expertise that is not always available to stakeholders. The development of more efficient preference elicitation and interaction procedures would increase the likelihood that decision-makers would make the extra effort required to use this potentially powerful method. Nonetheless, the mixed-methods approach had several important advantages over more traditional strategic planning methods including its ability to stimulate discussions amongst stakeholders who do not regularly interact, support collaborative planning, and encourage multi-benefit solutions.

1. Introduction

Researchers and practitioners are increasingly interested in improving urban water management by transitioning from existing segmented management approaches to integrated, multi-benefit approaches (Brown and Farrelly, 2009; Larsen and Gujer, 1997). Achieving this goal is socially, politically, and technically complex because water infrastructure affects many different stakeholders, lasts for multiple decades, and requires significant financial investment. Improved strategic planning processes can help facilitate this transition by allowing stakeholders to articulate their values and objectives, by considering innovative options, and by explicitly accounting for uncertainties about the future (Truffer et al., 2010). They also can support major shifts in water infrastructure investment by allowing decision-makers to consider the long-term benefits of potential systems in ways that are not captured by existing planning methods, which often result in incremental improvements (Dominguez et al., 2009).

To facilitate transitions to multi-benefit water infrastructure, decision-makers must engage with stakeholders who have historically been excluded from the decision-making process, for example, coastal land managers in the case of planning for wastewater infrastructure (Pearson et al., 2010). Researchers have developed qualitative strategic planning processes in which stakeholders describe uncertainties and qualitatively explore trade-offs amongst different management alternatives (Störmer

* Corresponding author. Energy and Resources Group, 310 Barrows Hall, University of California, Berkeley, 94720, United States.
E-mail address: sharrislovett@berkeley.edu (S. Harris-Lovett).

https://doi.org/10.1016/j.jenvman.2018.11.112
Received 18 June 2018; Received in revised form 1 November 2018; Accepted 23 November 2018

et al., 2009; Störmer and Truffer, 2009). This can include analysis of the social dynamics and long-term goals of stakeholders (Dominguez et al., 2011). Yet researchers suggest that decision-makers whose choices implicate large sums of money or influence many peoples' lives should use both qualitative and quantitative data to inform decision-making (Mays et al., 2005; Pope and Mays, 1995; Munda et al., 1994). In particular, transitions towards more sustainable urban water infrastructure are best supported by qualitative analyses of actors, networks, and institutions paired with scientific modeling, because decision-making needs to consider both social and physical aspects of the available options (Fratini et al., 2012). Therefore a mixed-methods approach is useful (Greening and Bernow, 2004).

Due to the long design lifetimes of most water infrastructure, considering future conditions is essential for decision-making. Scenario planning includes critical uncertainties about the future when analyzing potential management options, allowing decision-makers to explicitly consider a range of possible future conditions. It is becoming more popular among water infrastructure planning professionals (Kang and Lansey, 2012; Lienert et al., 2006) as well as within environmental management (Mahmoud et al., 2009; Peterson et al., 2003).

In addition to scenario planning, multi-criteria decision analysis (MCDA) can also guide environmental management (Huang et al., 2011; Liu et al., 2008; Reed, 2008; Mendoza and Martins, 2006; Linkov et al., 2006), water resources management (Kunz et al., 2013; Mutikanga et al., 2011; Marttunen and Hämäläinen, 2008; Gregory et al., 2006; Hajkowicz and Collins, 2006; Borsuk et al., 2001), and water infrastructure development (Garrido-Baserba et al., 2016; Zheng et al., 2016; Scholten et al., 2015; Kabir et al., 2014; Lienert et al., 2014, 2006; Al-Kloub et al., 1997). MCDA creates a structured framework for multiple objectives articulated by local stakeholders (Belton and Stewart, 2002; Keeney, 1992). Participating in MCDA interviews can help decision-makers clarify their own objectives for any given decision (Marttunen and Hämäläinen, 2008; Lichtenstein and Slovic, 2006; Gregory et al., 2001; Payne et al., 1992). This clarity is especially important for infrastructure projects because it promotes transparency in uses of public funds (McDaniels et al., 1999). By identifying the topics of greatest (dis-)agreement amongst stakeholders, MCDA can help avoid later conflicts (Hajkowicz, 2008; Hermans et al., 2007).

An MCDA framework effectively parses out the relative importance of stakeholder preferences from assessment of the technical performance of management options (Marttunen and Hämäläinen, 2008; Bojórquez-Tapia et al., 2005; Matsatsinis and Samaras, 2001). This distinction focuses discussion on objectives of the projects ("what is important") rather than on discrete management options ("how to achieve objectives") (Greening and Bernow, 2004; Lai et al., 2002). This facilitates considering innovative options that would not result from incremental improvements to existing infrastructure. Specifically, to create an overall ranking of options, MCDA helps identify the tradeoffs between achievement of objectives among different options (Gregory et al., 2001). This allows decision-makers to explicitly weigh trade-offs between social, cultural, environmental, and economic factors (Kiker et al., 2005).

Combinations of MCDA with other strategic planning methods (i.e., mixed-methods approaches) expand research in natural resources management (Kangas et al., 2002). Here, we combine stakeholder and scenario analysis with MCDA to provide decision support and facilitate strategic planning for nutrient management in the San Francisco Bay Area. This study extends previous mixed-methods approaches that have considered a range of uncertain future conditions, like population growth and climate change in multi-criteria decision analysis for water infrastructure planning (Zheng et al., 2016; Lienert et al., 2014). This mixed-methods approach is more broadly relevant for environmental management decisions characterized by problems that involve many stakeholders, require regional solutions and involve significant capital investments for long-term infrastructure.

1.1. Case study: nitrogen management in the San Francisco Bay

Nitrogen pollution adversely impacts many estuaries and coastal waters (Howarth, 2008; Howarth et al., 2002, 2000). Excess nutrients can cause oxygen depletion and eutrophication, reduce fisheries productivity and decrease recreational value (Dodds et al., 2008). Excessive nutrients discharges to surface waters can also result in growth of algae that exude harmful toxins (Heisler et al., 2008; Anderson et al., 2002; Van Dolah, 2000). Nitrogen pollution is complex, with context-dependent solutions, and has been termed a "wicked" problem (DeFries and Nagendra, 2017; Thornton et al., 2013).

Nutrient management in the San Francisco Bay Area exhibits many characteristics of complex environmental management decisions that are well-suited for a mixed-methods approach. Decision-makers must balance concerns about human effects on ecosystems, costs, and numerous human interactions (Benda et al., 2002). Scientific uncertainty, differing stakeholder opinions, and the need for regional solutions complicate decision-making (Balint et al., 2011).

The San Francisco Bay ecosystem has historically been insensitive to nitrogen pollution, likely because algal growth was light-limited due to suspended sediment associated with nineteenth century hydraulic mining and twentieth century water diversion projects (Cloern, 1999; Alpine and Cloern, 1988; Cole and Cloern, 1984). As the turbidity of the Bay declines and nitrogen levels increase with population growth (Cloern and Jassby, 2012), evidence indicates that algal growth is shifting from light-limited to nitrogen-limited (Cloern, 1999; Boynton et al., 1982). The extent of the impacts of nitrogen pollution are uncertain, especially when considering the effects of climate change and invasive species (Sutula and Senn, 2015).

Presently, most of the nitrogen entering the San Francisco Bay estuary ("the Bay") is associated with urban and agricultural runoff and the discharge of municipal wastewater treatment plants (Novick and Senn, 2014; Wankel et al., 2005; Hager and Schemel, 1992). South of the Bay Bridge, discharge of municipal wastewater effluent comprises more than 90% of the anthropogenic nitrogen load (Novick and Senn, 2014) (map in Supplemental Information, Fig. S1).

In response to the potential for future regulations to control nitrogen loads (San Francisco Bay Regional Water Quality Control Board, 2014), a varied group of stakeholders have begun to plan strategies to lower nutrient loads to the Bay. The Nutrient Management Strategy team is advised by a steering committee, a stakeholder advisory group, a technical working group, and a science team (San Francisco Bay Nutrient Management Strategy, 2016).

Nutrient pollution has been viewed traditionally as an issue with two sets of stakeholders: regulators and nutrient dischargers. Yet reality in the Bay Area is more complex, due to a strong interest in the region for providing nitrogen control infrastructure that also provides other benefits. These co-benefits include increased shoreline habitat, recreational shoreline access, water supply, and resilience to sea level rise (Harris-Lovett et al., 2018). Therefore, strategies to manage nitrogen loads may also affect other stakeholders like water supply managers, baylands land managers, and ecological stewards.

Unlike situations in which MCDA has been employed to find a single optimal solution (e.g. choosing a location for an airport; Bojórquez-Tapia et al., 2005), the diverse set of stakeholders in the San Francisco Bay must make a series of separate decisions that will advance collective goals. Wastewater treatment plant managers (dischargers) must preserve water quality while respecting the financial limitations of treatment plant ratepayers (Harris-Lovett et al., 2018). Dischargers must each decide on technologies or actions (if any) to control the mass of nutrients released in their effluent. Regulators must enforce laws and policies to protect the Bay ecosystem (e.g., Federal Water Pollution Control Act, 33 U.S.C. 1251-1376), so their decisions entail whether and how to set legal limits on nutrient loading. Other stakeholders, like baylands stewards, coastal planners, or environmental advocates, can decide whether to contest such decisions through litigation.

Within this varied decision-making context, MCDA on its own is not sufficient for providing regional strategic planning support. This issue, like other "wicked" problems of ecosystem management, requires decision support tools to facilitate multi-sector decision-making, enable collaborative decision-making across agencies and administrative boundaries, and balance different stakeholder values (DeFries and Nagendra, 2017). Hereby, strategic planning endeavors to facilitate greater understanding and teamwork amongst diverse stakeholders without necessarily aiming for consensus or a one-size-fits-all solution.

To provide this support, we combined stakeholder analysis and scenario planning with multi-criteria decision analysis. We evaluate the insights derived from this mixed-methods approach and generalize its applicability to strategic water infrastructure planning and management of complex environmental problems. To support decision-making about nutrient management in San Francisco Bay, we addressed the following specific aims:

1. Identify the objectives on which stakeholders agree and disagree, and clarify key areas where consensus should be achieved before decision-making proceeds.
2. Assess the value of different nutrient management options with respect to stakeholder objectives and preferences.
3. Define and bound the uncertainties associated with technical performance and future conditions for each management option. Determine which options perform most robustly under a range of future scenarios and across stakeholder viewpoints.
4. Determine areas in which further scientific research would be helpful in informing decision-making.

2. Materials and methods

2.1. Methodological overview

The research proceeded in several stages. (1) First, we selected stakeholders for participation in interviews. Initial interviews included a broad array of 32 stakeholders. Since follow-up interviews required a significant time investment from the participants and because it would be prohibitively complex to present 32 stakeholder results in an MCDA, we used (2) cluster analysis to group stakeholders. This cluster analysis ensured selecting stakeholders with high relevance to decision-making and representing a broad range of diverse viewpoints; the two most important aspects for participation in a (3) follow-up MCDA preference weighting interview. Data from the second interviews about stakeholder preferences and ideas for potential management solutions informed (4) a multi-criteria decision analysis to determine the relative value of various management options. Stakeholder interview data also informed (5) future scenarios, which were (6) paired with MCDA to determine ways in which management options' value changed under uncertain future conditions. Each step of the research process is detailed below.

2.2. Stakeholder selection

Stakeholders were identified based on their professional interest and expertise in nutrient loading to San Francisco Bay, specifically whether they were involved with decision-making or would be affected by decisions made (Reed et al., 2009; Grimble and Wellard, 1997). Stakeholders were chosen to represent diverse groups of regulators, wastewater managers, coastal stewards, researchers, and advocates (for environmental or industrial causes). These stakeholders go far beyond the dischargers and regulators who historically have made decisions about wastewater infrastructure. Although wastewater infrastructure affects all people in the region, wastewater has typically been considered "out-of-sight, out-of-mind" by the general population in California (Harris-Lovett and Sedlak, 2015). For this reason, only those with professional (not personal) interest were chosen as stakeholders, resulting in selection of experts and selected representatives of stakeholder groups.

Stakeholder identification proceeded in three iterative stages:

1.) We identified organizations and individuals involved with decision-making about nutrient management as evidenced by their presence on relevant advisory committees (e.g., Nutrient Management Strategy), by appearances at relevant public meetings (records of attendees), or by authorship of relevant documents. When an organization (and no particular person within it) was identified, the person within the organization with the most responsibility for strategic planning was contacted using publicly-available professional email addresses and asked to participate or to recommend someone within the organization to participate.
2.) Once interviews commenced (see Section 2.3), we used snowball sampling (Atkinson and Flint, 2001; Biernacki and Waldorf, 1981) to identify other stakeholders. Additionally, participants rated their own influence over decision-making as well the extent to which decisions about nutrients would affect them, on a scale of 1–7. They also rated the influence and defined the extent to which others would be affected (see approach in Lienert et al., 2013). Multiple stakeholders from a single organization were contacted when they had distinct roles in decision-making and when they were specifically identified by other stakeholders.
3.) The researchers determined stakeholders who would be affected by proposed nutrient management options, which were described in regional planning documents and discussed in initial interviews. For example, references to upgrading treatment plants to include biological nutrient removal technologies (e.g., San Francisco Bay Regional Water Quality Control Board, 2014), resulted in including stakeholders from engineering consulting companies who would conduct the work. Constructing wetlands for nutrient removal (e.g., Wren, 2017) resulted in including coastal land managers.

2.3. Initial interviews

Initial interviews were designed to collect data for:

1 Conducting a stakeholder analysis that illuminated the history and current state of decision-making about nutrients.
2 Building an objectives hierarchy for the MCDA.
3 Defining attributes for the decision criteria in the MCDA.
4 Developing ideas for nutrient management options.
5 Determining critical future uncertainties to test in the MCDA.
6 Increasing understanding of how nutrient management fit into other long-term planning objectives for the estuary and into stakeholders' professional mandates (see Harris-Lovett et al., 2018).

Initial interviews were semi-structured. Open-ended questions were designed to elicit stakeholders' interest in nutrient management in San Francisco Bay and their role in decision-making. We elicited objectives for good nutrient management and ideas for ways to measure fulfillment of these objectives ('attributes' for MCDA). We also elicited ideas for potential nutrient management options and future conditions that might affect nutrient management (see interview guidelines in Supplemental Information, Fig. S2).

First round interviews lasted 30–90 min and were conducted by telephone, with the exception of four individuals from one organization who were interviewed in person together. Prior to their interview, these four individually completed handouts to elicit individual points of view; the entire group then engaged in discussion. All interviews were transcribed and coded for themes pertaining to the aims of the interview (listed above) using MaxQDA software, version 12.3.2. The software was manufactured by VERBI Software GmbH, based in Berlin, Germany, and purchased online at www.maxqda.com.

The interview protocol was approved by the Committee for

Protection of Human Subjects (the Institutional Review Board) at the University of California, Berkeley.

2.4. Development of objectives hierarchy and attributes

Synthesizing information from the first-round interviews, we developed a hierarchy of fundamental objectives (Eisenführ et al., 2010) for decision-making about nutrient management in San Francisco Bay, along with attributes to measure each objective. A top-level fundamental objective served as an umbrella for a similar number of fundamental sub-objectives (2 or 3). This balanced grouping in hierarchy branches minimized splitting biases (Hämäläinen and Alaja, 2008). Overarching goals for good nutrient management were informed by previous MCDA analysis of water infrastructure planning (Lienert et al., 2014).

Attributes were chosen to measure how well different management options fulfilled each objective. We attempted to choose attributes that directly related to the objectives, that could reasonably be determined for each option, and were understandable, comprehensive, and unambiguous. These characteristics are best-practice for attribute selection in MCDA (Eisenführ et al., 2010). However, these best practices for attribute selection could not always be fulfilled; in some instances there was no clear consensus from stakeholders on how to measure fulfillment of an objective. This necessitated selecting an attribute informed by consultation with specialists. One example was "good water quality": some stakeholders stated that this objective should be assessed by probability of impairment, others indicated that it would be met only in the absence of impairment, and others indicated that a proxy measure, like abnormally low concentrations of dissolved oxygen or high concentrations of chlorophyll-a would indicate a failure to meet the objective. After consulting with several water quality experts, we chose the attribute of probability of impairment of water quality. For "maximizing removal of contaminants of emerging concern from wastewater effluent," which encompasses a wide range of different chemical compounds, we chose the proxy attribute of mass loading of the antibiotic sulfamethoxazole because it was well-characterized in the literature (Jasper et al., 2014a; Jasper and Sedlak, 2013; Radjenović et al., 2008; Batt et al., 2007). We expressed attributes in continuous scales only when we deemed no other attribute to be appropriate. One example was the objective of ease of adaptation which ranged from 0% (impossible to adapt to changing conditions) to 100% (very easy and cheap to adapt to changing conditions).

2.5. Development of management options

Potential options to manage nutrients were derived from stakeholder interviews, technical documents (e.g., permits, see San Francisco Bay Regional Water Quality Control Board, 2014), and informed by our own understanding of management approaches (see below). The latter included 'Do nothing' to provide a baseline for comparison, as well as urine source separation and treatment as an approach unfamiliar to the local decision-makers but having been considered a viable option in Europe (Lienert and Larsen, 2007). Brief descriptions of management options are in Table 1 (detailed descriptions in Supplemental Information, Table S1). The nitrogen loading reductions below 2017 levels were calculated based on model assumptions (detailed in Supplemental Information, Text S1).

We applied each option to the entire case study region to assess the extent to which it could meet different objectives. However, it is important to note that this approach is not realistic; decisions about technology adoption are much more likely made at a local scale to fit specific needs related to existing infrastructure, local geography and institutional constraints. The MCDA results for the management options should be considered instructive and illustrative but not prescriptive.

Most management options were developed with input from stakeholders in initial interviews. However, the following three management options were added after follow-up interviews with stakeholders to assess how different permutations of the original options affected final rankings:

1.) Constructed open-water wetlands were added as a comparison with sub-surface flow "horizontal levee" wetlands, with different assumptions about land availability, wetland sizing, costs, habitat creation, resilience to sea level rise, and nutrient removal.
2.) Urine source separation initially focused on "early adopters" of the technology. A second option was added with financial incentives for adoption of urine source-separation technology, which would achieve greater levels of nutrient removal and would increase reliability of this option.
3.) Potable water recycling with a line for discharging reverse-osmosis concentrate to the ocean was added to address stakeholders' interest in potable water reuse as a water supply option while still reducing nutrient loading to the Bay.

Each option was developed by considering the maximum reasonable extent to which it could be applied in the region, based on high-end estimates obtained in planning documents, scientific literature, and from conversations with stakeholders. As a result, each option represents different levels of nutrient removal and different degrees of fulfillment of each objective. All assumptions and detailed parameters of each management option modeled in the MCDA are included in the Supplemental Information, Text S1.

2.6. Future uncertainty

To assess the effect of uncertainty in future conditions on technological options for nutrient management, we considered several key factors in the year 2050. We chose 2050 because most wastewater infrastructure lasts at least 30-years (Dominguez and Gujer, 2006).

During interviews, stakeholders listed future conditions in 2050 that would likely affect their choice of nutrient management options ("critical uncertainties"). By definition, these were outside of the control of water managers but would profoundly impact the choices of interviewees (Wilkinson and Kupers, 2014). We distilled this information into factors that would most likely influence MCDA results (Mahmoud et al., 2009): population growth, effects of climate change, and the Bay's ecological resilience with respect to nutrient loading. The latter would affect "good water quality" for all treatment options. Changing population size could affect nutrient loading (and hence water quality), loading of contaminants of emerging concern, and sizing of treatment options (which would indirectly affect greenhouse gas emissions and cost). Climate change-related impacts (e.g., magnitude of sea level rise) could affect resilience of treatment options to sea level rise.

Two of these critical uncertainties (i.e., nutrient loading affected by population change and the Bay's ecological resilience to nutrients) were used to develop a matrix of possibilities with which to inform the development of future scenarios (Scott et al., 2012; Wright and Goodwin, 2009) (Supplemental Information, Fig. S3). The effect of climate change on wastewater infrastructure located at or near sea level was used to amplify the Worst- and Best-case scenarios developed in the matrix. Population change assumptions were made by extrapolating the most rapid rates of growth and decline in the Bay Area in the previous 30 years (Association of Bay Area Governments, 2016) out to 2020. Effects of sea level rise were modeled by assuming a 10% decrease in the score for the objective "resilience to sea level rise" for each level of increase in effects of sea level rise (on a scale from 0 to 5).

Our scenarios identify extreme futures by placing positive elements for nutrient control in one scenario and negatives in another (Schoemaker, 1995):

- **Worst-case scenario for nutrient impairment.** In this scenario, the Bay's ecosystem is more sensitive to nutrients due to ecosystem

Table 1
Nitrogen management options considered in the multi-criteria decision analysis. (Photo credits: Wastewater treatment– By Hasan Zulic/panoramio/CC BY 3.0/Wikimedia Commons; Wetlands – By US Fish and Wildlife Service/Wikimedia Commons; Recycling – CC-BY-SA3.0/Wikimedia Commons; Roediger NoMix Toilet (urine-separating toilet) – By Sustainable Sanitation Alliance Secretariat/CC BY 2.0/Wikimedia Commons).

Management option		Description	Nitrogen loading reduction below 2017 levels
Do nothing		No additional action.	0
Constructed wetlands 	Horizontal levee wetlands	Vegetated wetland levees are built to the maximum possible extent given spatial constraints.	53%
	Shallow open-water wetlands	Open-water wetlands are built to reduce nitrate loads by 90% at each wastewater treatment plant if possible, given spatial constraints.	65%
Wastewater recycling 	Increase recycling for irrigation	Maximize wastewater recycling for irrigation (without additional treatment for nutrient removal).	28%
	Increase recycling for potable reuse	Maximize recycling of wastewater for potable reuse, with a "brine line" to the ocean, thus diverting nutrients from the Bay.	26%
Urine source-separation and treatment 	Install urine source-separating toilets–early adopters	Deploy urine-separating toilets in all new housing and for some early adopters to divert and treat urine in decentralized facilities.	1%
	Install urine source-separating toilets– with incentives	Deploy urine-separating toilets in new housing to divert and treat urine in decentralized facilities, with financial incentives to encourage 30% adoption in existing housing.	14%
Wastewater treatment plant upgrades (as per HDR report specifications) 	Optimization	Optimize existing wastewater treatment processes for total nitrogen removal.	10%
	Level 2 upgrades	Upgrades to achieve < 15 mg TN/L.	55%
	Level 3 upgrades	Upgrades to achieve < 6 mg TN/L.	82%

attributes like decreased water column turbidity and increased stratification periods. Nutrient loading to the Bay increases by 60% due to rapid population growth between 2017 and 2050. Climate change strongly affects the performance of existing wastewater treatment plants.

- **Best-case scenario: less pressure for nutrient control.** In this scenario, the Bay retains a strong resilience to nutrient pollution. Nutrient loading to the Bay decreases by 13% due to population decline between 2017 and 2050. Sea level rise does not affect existing wastewater treatment systems.

The "status quo" scenario assumes 33% population growth by 2050 (roughly 1% per year; Association of Bay Area Governments and the Metropolitan Transportation Commission, 2017), no effects of climate change on wastewater treatment, and increased ecological sensitivity to nutrient loading. Additionally, we designed the model so that attribute values could be calculated for any level of population size change, five levels of climate change effects between these two extremes, and with or without increased ecological sensitivity to nutrient loading (see associated document, 'R code for SI and sharing' and Supplemental Information, Text S2).

The model was run in the open-source software R (R Core Team, 2013), primarily using the package 'utility' for the MCDA, as well as other packages for analysis and presentation of data ('fitdistrplus', 'truncnorm', 'RColorBrewer', and 'plyr') (Delignette-Muller and Dutang, 2015; Neuwirth, 2014; Trautmann et al., 2014; Reichert et al., 2013; Wickham, 2011). Open-source software was deliberately chosen to allow stakeholders and other researchers to conduct the MCDA under a range of future conditions.

The simulations of uncertainty included in the MCDA were based on 1000 model runs for ease of computing. Although previous MCDA studies which included analyses of uncertainty used 10,000 model runs (Zheng et al., 2016), comparison of median overall values with 1000 compared to 10,000 were similar and did not change the ranking order of any alternative under any future scenario.

2.7. Stakeholder analysis and selection for follow-up interviews

We conducted follow-up interviews with a subset of the original group to elicit a range of opinions on the relative importance of the objectives for nutrient control. MCDA results depend on the preferences of decision-makers. Although numerous decision makers exist in this case (many more than the 32 initially interviewed), MCDA results in regional decision settings are useful if they can clarify the broad range of interests at play. Here, the selection of participants was designed to represent the breadth of opinions amongst the stakeholders because initial interviews suggested that individual stakeholders with outlier opinions could have an outsize role in affecting the decision-making process through litigation or negative media attention. To sample these differences of opinion, we performed a cluster analysis of the 32 initially interviewed stakeholders (Mardle et al., 2004; Zahir, 1999). We categorized each response based on stated goals for nutrient management (presence/absence of each objective in stakeholders' answers to questions about goals for nutrient management).

Our methodology was derived from statistical methods in community ecology. We chose cluster analysis because it serves our purpose to cluster similar observed characteristics within a larger group (Borcard et al., 2011; McCune and Grace, 2002). We used the software 'R' with packages 'vegan', 'cluster', 'indicspecies' and 'permute' (Oksanen et al., 2017; Simpson, 2016; Maechler and Hornik, 2016; R Core Team, 2013; Dufrene and Legendre, 1997). To form the clusters, we excluded mentions of the objective "good water quality", because it was clear from the other interview questions that many stakeholders who had not specifically mentioned "good water quality" as a goal implicitly assumed it was a high priority. We also removed a stakeholder who did not name any objectives for "good nutrient management" due to interview time constraints.

We used a Bray-Curtis distance to form the clusters (which clusters only on shared presence, not shared absences) (Zuur et al., 2007) to group stakeholders by the objectives they considered most important. We used a flexible-β linkage, with parameters $\alpha 1 = .625$, $\alpha 2 = .625$, $\beta = -0.25$, $\gamma = 0$ to determine the optimal size and shape of each cluster. We then used a Mantel Test to prune the dendrogram formed in the cluster analysis (Borcard et al., 2011). This resulted in seven clusters, with one to eleven stakeholders. We also conducted a statistical analysis to determine which objectives within each cluster of stakeholders most differentiated them from the other clusters (called an "indicator species analysis" in ecology) (Dufrene and

Table 2
Stakeholder clusters based on stated goals for nutrient management. Stakeholders 1–9 (in bold) participated in follow-up interviews. Relevance denotes how strongly a stakeholder was engaged in or affected by decision-making about nutrient loading (1 = directly involved in decision-making; 2 = strongly affected by decision-making, or with strong influence over decision-makers; 3 = slightly affected by decision-making; 4 = interested/concerned with nutrients, but not directly affected by decision-making).

Objective cluster group	Cluster group characteristic	Relevance	Stakeholder	Professional role
1	Wildlife habitat	**1**	**SH6**	**regulator**
		1	SH12	regulator
		2	SH19	discharger
		2	SH16	regulator
		2	SH18	researcher, advocate
		3	SH25	steward, researcher
2	Low costs and water supply	**1**	**SH8**	**advocate**
		1	SH22	discharger
		1	**SH2**	**discharger**
		1	**SH9**	**regulator**
		1	SH10	regulator
		2	SH21	discharger
		2	SH23	engineer
		3	SH26	water supplier
		4	SH17	advocate
		4	SH24	researcher
		4	SH15	water supplier
3	Need science-based understanding of nutrient effects on ecosystem	**1**	**SH1**	**advocate**
		1	SH32	discharger
		1	**SH3**	**discharger**
		1	**SH4**	**regulator**
		1	SH13	regulator
		2	SH27	regulator
4	Technical reliability	2	SH30	discharger
		4	SH11	planner
5	Collaboration across professional fields	1	SH28	discharger
		1	**SH7**	**discharger**
		4	SH31	engineer, planner, regulator
		4	SH20	researcher, steward
6	NA	4	SH14	regulator
7	Balance nutrients with other long-term management goals	**3**	**SH5**	**steward**
		4	SH29	engineer, planner, regulator

Legendre, 1997). These results are depicted in Table 2.

From each of these clusters, we contacted those stakeholders with the highest relevance to decision-making (classified on a scale of 1–4, with 1 being most engaged with or most affected by decision-making about nutrient loading). Further selection criteria for follow-up interviews included individuals who had the greatest interest in nutrients in the southern reach of the Bay as determined from initial interviews, and those with diverse professional roles in different agencies. We aimed to include at least one stakeholder from each cluster group in follow-up interviews. Of the 10 stakeholders contacted to participate in follow-up interviews, 9 agreed to participate (stakeholder 30, an engineer at a municipal wastewater utility, was contacted to participate in a second interview but this person had left their job and was not available).

We randomly assigned numbers 1–9 to the stakeholders who participated in the second-round interviews, and numbers 10–32 for stakeholders who participated in the first-round interviews only.

2.8. Follow-up interviews for preference elicitation

In follow-up interviews with nine selected stakeholders, we elicited weights using the Swing method (Schuwirth et al., 2012; Mustajoki et al., 2005). It was chosen because it has been previously used for decision-making about water infrastructure planning (Zheng et al., 2016). Swing is one of the most popular and well-established MCDA weight elicitation methods (see textbooks, e.g. Eisenführ et al., 2010; Belton and Stewart, 2002). To do so, stakeholders first read descriptions of each objective printed on notecards (see Supplemental Information, Text S4 and Text S5) and discussed the objective's importance. They started the weighting exercise by considering a “hypothetical option” where all objectives would be on their worst levels (receiving 0 points). Stakeholders then determined the most important objective to improve from the worst to the best value, assuming that all other objectives would remain on their worst levels; this “best hypothetical option” was assigned 100 points. Stakeholders then chose the next-most important objectives to improve from worst to best values and assigned points (0–100) to each.

Relative point values were then cross-checked for consistency across the objectives hierarchy with stakeholder feedback and adjustments where necessary (Belton and Stewart, 2002). Assigned points were normalized into weights on a scale of 0–1 for each objective for each stakeholder (Belton and Stewart, 2002). Per definition, the sum of weights for each stakeholder equals 1.

Stakeholders were asked to explain their rationale for assigning points to provide insight into their perspectives on the importance of the objectives and the suitability of the attributes (Marttunen et al., 2015). Attempts to confirm weightings from point allocation results with another common weight elicitation method, the trade-off method (Eisenführ et al., 2010), were almost uniformly rejected by stakeholders (see discussion in Section 4.3.4).

For the objectives that received the highest weights (and others if time allowed), we elicited the shape of the single-attribute value functions (i.e. whether improvement from the worst to best case fulfillment of the objectives was linear, concave, or convex). We used the bisection elicitation method (Eisenführ et al., 2010). If information was missing, we assumed linear value functions. We also identified any thresholds below which everything was equally bad or above which everything was equally good (Scholten et al., 2015). Interview guidelines for follow-up interviews are in the Supplemental Information, Fig. S2. A more detailed description of methods for preference elicitation is included in Supplemental Information, Text S3.

Second round interviews were conducted in person and took 60–120 min.

Protocols for follow-up interviews were approved by the Human Subjects Committee (the Institutional Review Board) at the University of California, Berkeley.

2.9. Prediction of attribute values for each management option

Predictions of attribute outcomes are uncertain, especially in complex environmental systems (Reichert et al., 2015). We used a combination of estimates from the literature, expert assessment, and modeling to determine an uncertainty range for each attribute prediction (Scholten et al., 2013).

Given the estimated uncertainty range and distribution for each attribute value (see Supplemental Information, Table S2), we developed a matrix of 1000 random potential attribute values for each objective for each option. If less than 3% of modeled values fell outside the worst-best range (as in the case of a normal distribution with a mean of 98 and a standard deviation of 1, with a top limit of 100), the mean value was used to replace those values that exceeded the limits of the range so that one extreme would not be over-represented in the model.

After calculation of the attribute values for each option, the option of potable water recycling with a pipeline to the ocean to dispose of reverse-osmosis brine was found to have a value of CO_2 emissions two orders of magnitude higher than the “worst” value used in the elicitation process. (This option was added after the second interviews based on stakeholder interest in potable water recycling as a means of nutrient control.) Some MCDA practitioners have suggested that the attribute range can be extended by assuming that stakeholders' preference weights would increase linearly (Eisenführ et al., 2010). However, this assumption is likely invalid in this case given how far outside the initial range this new option lies. Even with the un-adjusted weightings, potable water recycling with a pipeline for brine disposal option scored relatively low for most stakeholders (Supplemental Information, Table S6). Moreover, because the re-adjusted weights within the new range of CO_2 values would dramatically increase the weight of CO_2 emissions (and thus decrease the option's overall score), we decided not to include this option in the remainder of the analyses. To include it, either the stakeholders' objective weights for such high potential CO_2 emissions would have to be re-elicited, or the option would have to be reconfigured with another means of concentrate disposal (e.g., zero-liquid discharge systems, emerging concentrate treatment technology).

2.10. Multi-criteria decision analysis

The MCDA was conducted in R using the 'utility' package (Reichert et al., 2013). The attributes for the objectives (termed 'end-nodes' in the MCDA software) were assumed to be single-attribute continuous parametric functions (for each value of 'x' there is only one value of 'y'). Overarching objectives (mid- and top-level aggregation nodes) were assumed to be aggregations of lower-level nodes and were assumed to convert to overall values using a continuous parametric function (rather than discrete classes).

Two stakeholders required separate objectives hierarchies. For stakeholder 3 (SH3), an objectives hierarchy was built that did not include water quality because the stakeholder refused to consider a probabilistic measure of impairment. Instead, SH3 insisted that impairment should be a "true/false" measure of existing ecological conditions. SH3's objectives hierarchy also did not include habitat because SH3 refused to choose between personal and professional sentiments about the importance of habitat regarding nutrient management (SH3 indicated that their personal opinions were at odds with their professional mandates). For stakeholder 7 (SH7), the objectives hierarchy did not include recovery of nutrients from wastewater, because SH7 refused to accept a measure of nutrient recovery that did not include recovery from solids removed during conventional wastewater treatment. Though SH3 and SH7 are included in the results, their overall rankings of options, while still instructive, are not comparable to those of other stakeholders.

We used a simpler MCDA model to determine results for each stakeholders' individual preferences (additive aggregation, linear value functions, no uncertainty in stakeholder weights) because our model aims to provide structure for discussion within a broader decision-making effort, not to definitively provide a solution to a problem (Scholten et al., 2017).

2.10.1. MCDA aggregation function

As a baseline case, we assumed additive aggregation of the nodes to determine the overall value of each option. Additive aggregation is commonly employed in MCDA. It implies that high attribute values for one objective completely compensate for low attribute values for another (Eisenführ et al., 2010). However, additive aggregation has been shown to be an inaccurate representation of stakeholder perspectives in some cases (Langhans et al., 2014). Despite this shortcoming, additive aggregation is a valid simplification in some MCDA cases because changes in the aggregation method do not necessarily change the option ranking in the MCDA output (Scholten et al., 2017).

2.10.2. Marginal value functions

As a baseline case, all value functions were assumed to be linear and without thresholds (no strict limits in attribute values). Linear value functions are a valid simplifying assumption for many MCDAs (Scholten et al., 2017). We resorted to these because time constraints in interviews prevented us from querying all interviewees about the shape of the marginal value function of attribute fulfillment or thresholds. Additionally, most interviewees who discussed value functions gave vague curvatures rather than discrete midpoint values. The baseline case assumptions and resulting rankings were then tested in a sensitivity analysis (sections 3.5.1 and 3.5.4) (Scholten et al., 2017; Zheng et al., 2016).

3. Results

3.1. Stakeholder analysis

The selected group of stakeholders represented 76 organizations, including water managers, baylands ecological stewards, scientific researchers and engineers, regulators, urban planners, flood control managers, and advocates for coastal industry or environment at the local, regional, and federal scales (Kunz et al., 2013). Several stakeholders represented more than one organization (e.g., one person served as director of an industrial advocacy group and also served on the board of a public wastewater utility). Of the 88 individuals contacted, 32 stakeholders (representing 29 different organizations) participated in an interview.

Stakeholders with the same professional role (i.e. discharger, regulator) and even within the same organization frequently stated different goals for nutrient management, as evidenced by their failure to cluster together (Table 2). They also weighed the importance of objectives differently. In other words, it would be inaccurate to assume that all dischargers or all regulators have the same objectives. Stakeholder weights for each objective are in the Supplemental Information, Fig. S4.

Several stakeholders mentioned thresholds for low fulfillment of certain objectives. Three respondents said they would not accept any option for nutrient management that was below a certain probability (%) of deviating from good nutrient-related conditions in the southern reach of the Bay. These probabilities were worse than 15% (wastewater dischargers advocate), 20% (coastal land steward), and 50% (regulator). One stakeholder (regulator) would not endorse any nutrient management option that did not protect existing infrastructure from sea level rise. Two stakeholders (coastal land steward, wastewater dischargers advocate) would not accept any option with levels of ease of adaptation below 76% and 50%, respectively. Three stakeholders (regional regulator, coastal land steward, wastewater dischargers advocate) would not accept any option with levels of reliability below 70%, 80%, and 85%, respectively. Effects of these thresholds on the MCDA results are calculated in the sensitivity analysis (section 3.5.1).

3.2. Objectives for good nutrient management

The objectives and attributes for good nutrient management in San Francisco Bay (Table 3) indicate stakeholders have a wide variety of goals, some of which are outside the scope of traditional wastewater infrastructure planning (Harris-Lovett et al., 2018). For details and rationale about how each attribute was calculated, see the Supplemental Information, Text S1.

3.3. Prediction of attribute values

Mean attribute values for the Status Quo scenario are shown in Table 4 as an example. Attribute values for the Worst- and Best-case scenario, and status quo population growth without increased sensitivity to nutrient loading are in the Supplemental Information, Tables S3–S5.

Table 3
Objectives and attributes for good nutrient management in the San Francisco Bay, based on results from stakeholder interviews. (Photo credits: Heron fishing – Chris Harshaw/CC BY-SA 3.0/Wikimedia Commons; Wastewater treatment plant –OpenStax/CC BY 4.0/Wikimedia Commons; Footprint – from http://www.greencareers.biz/faq/what-does-it-mean-to-offset-your-carbon-footprint/; Thumbs up – Pratheeps/Wikimedia Commons).

Goal	Objective	Attribute	Unit	Description
Healthy estuarine ecosystem 	Good water quality	Probability of deviating from good nutrient-related water conditions	%	Nutrient over-enrichment could result in eutrophication and impairment of beneficial uses (Sutula and Senn, 2015). Expert estimates of the attribute were made based on percent nitrogen load change from current levels.
	Good wildlife habitat	Area of additional wetland habitat created	Square hectares	The modeled area of constructed wetland habitat was based on results from a previous analysis (Wren, 2017).
Maximize treatment and beneficial uses of wastewater 	Increase water supply	Usable water produced	MGD (million gallons/day)	Attribute estimates were derived from utility planning documents (e.g., San Francisco Public Utilities Commission, 2016).
	Increase resource recovery	Recovered nitrogen (N) from effluent that can be used as fertilizer	Kg N/Year	Attribute estimates for nitrogen recovery from urine-source separation were derived from academic literature (e.g., Tarpeh et al., 2017). Recycling for irrigation was assumed to utilize all nutrients (Vazquez-Montiel et al., 1996).
	Maximize removal of unregulated contaminants (CEC)	Total sulfamethoxazole (SMX) loading in the southern reach of the Bay	Kg SMX/year	Sulfamethoxazole was used as a proxy for CECs because its removal in wastewater treatment is relatively well characterized (Batt et al., 2007; Jasper et al., 2014a; Jasper and Sedlak, 2013; Radjenović et al., 2008).
Promote inter-generational equity 	Ease of adaptation as conditions change	Percent ease of adaptation (considers sunk costs, time, physical potential)	%	Wastewater infrastructure that can be quickly and cheaply adapted to deal with changing influent flows and/or concentrations and to achieve more stringent regulatory standards is desirable. Classification: 0–50%: Impossible or hard to adapt; 51–75%: Moderately adaptable; 76–100%: Easy to adapt.
	Resilience to sea level rise (SLR)	Extent to which technology is vulnerable to SLR and storm surges	Constructed scale	SLR poses a threat to many of the Bay Area's wastewater treatment plants (Heberger et al., 2009). Scale from −10 to 10; with 10: Protects existing assets from SLR; 0: Unaffected by SLR; −10: Highly vulnerable to SLR.
	Low greenhouse gas (GHG) emissions	Lifecycle GHG emissions of wastewater treatment	CO_2 eq./year	Attribute estimates from literature on emissions of wastewater treatment (Stokes and Horvath, 2009) and nitrogen removal (Corominas et al., 2013).
Good social support 	Maximize public's ease of use	Percent ease of use	%	Some technologies could require involvement or behavior change by end users (e.g., urine source-separating toilets requiring men to sit when urinating). Classification: 0–60%: Requires behavior change from users; 61–80%: Mental shift but no behavior change required; 80–100%: Easy to use.
	Increase shoreline access	Additional access points (above 2017 levels)	Number	Access to aesthetically pleasing places along the Bay shore for recreation is desirable.
	Maximize ease of permitting	Percent ease of permitting	%	Classification: 0–60%: Permitting requires much additional staff time and/or legislative change; 61–80%: Permitting requires some additional staff time; 81–100%: Easy to permit.
Minimize costs 	Technical reliability	Percent of time technology operates as intended	%	Derived from expert estimates.
	Minimize capital investment and O&M costs	Net present value over 30-year span	$	Cost calculations were based on initial capital investment costs and annual operation and maintenance costs (O&M) over a 30-year technology life span. No depreciation rate was used. If 30-year O&M costs were not available, current annual O&M cost estimates were assumed to remain constant over 30 years.

Table 4

Mean attribute values (rows; see Table 3) for each management option (columns; see Table 1) in 2050 assuming the Status Quo scenario: 33% population growth, no effects of climate change on wastewater operations, and increased ecological sensitivity to nutrient loading.

Objectives – Attributes [units]: best-worst range	Management options									
		Constructed wetlands		Wastewater recycling		Urine separation and treatment		Wastewater treatment plant upgrades		
	Do nothing	Wetland levee	Wetland open water	Recycle irrigation	Recycle brineline	Urine early adopters	Urine incentives	Optimization	Level 2 upgrades	Level 3 upgrades
Good water quality – Deviation probability from good quality [%]: 5-95	74	40	32	59	60	68	57	67	35	17
Good wildlife habitat – Additional wetland habitat [square hectares]: 0-5200	0	4200	790	0	0	0	0	0	0	0
Increase water supply – Usable water produced [MGD]: 154-22	22	22	22	120	100	22	22	22	22	22
Increase resource recovery – Nitrogen (N) recovery [Mkg N/year]: 8500-0	0	0	0	3300	0	1800	4300	0	0	0
Maximize removal of contaminants of emerging concern – Total sulfamethoxazole loading [kg SMX/year]: 31-137	85	56	70	60	64	56	47	86	78	79
Ease of adaptation – Percent ease of adaptation [%]: 100-0	100	53	42	45	45	85	55	75	52	10
Resilience to sea level rise – Scale [-10 (highly vulnerable) to 10 (protects infrastructure)]: 10 to 10	−5	8	5	−3	−5	0	0	−5	−5	−5
Low greenhouse gas emissions – Lifecycle emissions for wastewater treatment [Kkg CO_2 eq./year]: 200-900	290	350	310	430	70,000	290	290	290	380	660
Maximize public's ease of use – Percent public ease of use [%]: 100-0	100	100	100	100	55	33	35	100	100	100
Increase shoreline access – Number of additional access points [#]: 17-0	0	8.5	8.5	0	0	0	0	0	0	0
Maximize ease of permitting – Percent ease of permitting [%]: 100-0	100	33	31	80	45	45	40	90	90	90
Technical reliability – Percent of time nutrient technology operates as intended [%]: 98-50	100	77	77	91	98	66	76	98	98	98
Minimize initial capital investment and O&M costs – Net present value over 30-year span [$1,000,000]: 0–8000	0	2700	1200	2200	370	400	5300	170	2500	3200

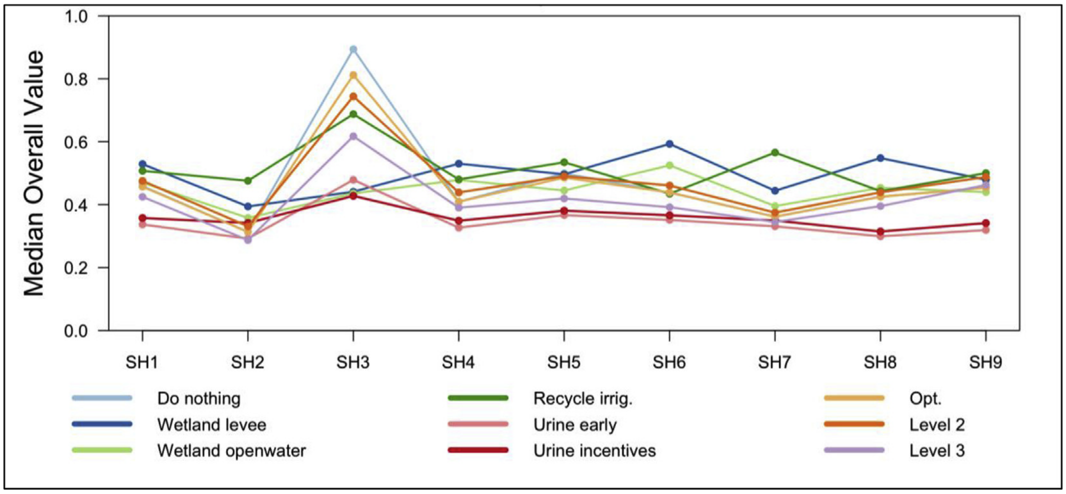

Fig. 1. Median overall MCDA value for each management option (colored lines; see Table 1) for each of nine stakeholders (SH; on x-axis) in the Status quo scenario. A value of 1 indicates that all objectives are fully achieved, a value of 0 that none of the objectives are achieved.

3.4. Multi-criteria decision analysis

The MCDA produced an overall value for each management option and stakeholder based on the attribute values and the stakeholder weights for each objective (Fig. 1). There was no option that scored highest for all stakeholders. However, increasing wastewater recycling for irrigation (in dark green) and building horizontal wetland levees (in dark blue) were among the top three options for most stakeholders under all future scenarios (Fig. S3 for other future scenarios). Conversely, both urine source-separation options (in pink and red) and Level 3 upgrades of wastewater treatment plants (in purple) were the lowest ranked options for most stakeholders under all future scenarios.

In the Best- and Worst-case scenarios, the median overall scores were strikingly similar to those of the Status quo, but the overall values were shifted slightly higher for the Best-case scenario and slightly lower for the Worst-case scenario (shown in Supplemental Information, Fig. S5).

Including uncertainty about attribute predictions into calculations of overall value for each option (Fig. 2) indicated that uncertainties in attribute predictions made more difference to overall value than future conditions for the less-established management options like the wetland options (horizontal levee and open water), recycling for irrigation, and the urine source-separation. In these cases, the values for each future scenario were very close (solid colored lines), while the values including the uncertainty of attribute predictions were far apart (dashed lines; see top rows (middle and right) and middle row in Fig. 2). Future conditions were the main cause of uncertainty in overall value for options in which the management option performance was well established, like optimization, Level 2, and Level 3 upgrades of wastewater treatment plants. In these cases, the dashed lines (uncertainty of attribute predictions) were close to the respective solid lines (median of scenarios), while the values for the scenarios (colors) differed relatively strongly (bottom row in Fig. 2).

Overall scores for each option for each stakeholder were converted to ranks (from 1 to 9, with 1 being the top-ranked option compared to the others). In this analysis, less well-established options like constructed wetlands (horizontal levees) and increased recycling for irrigation were likely to be in the top three ranked options for 8 of the 9 stakeholders in the Status quo scenario (Fig. 3). In many cases, the rank of each option was affected by uncertainty in attribute prediction, often spanning 4 or more ranks.

The probability of the top three ranked options for each stakeholder was virtually unchanged in the Worst-case scenario (Supplemental Information, Fig. S6). For the Best-case scenario, the 'Do nothing' option moved into the top three or became far better for nearly all stakeholders, and optimization improved for many stakeholders as well (Supplemental Information, Fig. S7). In other words, traditional responses fared better under the Best-case scenario. In general, Level 3 upgrades of treatment plants and the urine-source separation options ranked lower than the other options for nearly all stakeholders under a range of future conditions (Supplemental Information, Figs. S8–S10).

A closer look at the contribution of each objective to overall values for an individual stakeholder reveals that the benefits other than water quality of some of the less-traditional nutrient management options helped boost their overall value above that of conventional wastewater treatment plant upgrades. For example, for stakeholder 4 (SH4), the three best-performing options (two wetland options, recycling wastewater for irrigation) achieved comparatively high values on nearly all objectives (Fig. 4). Notably, the wetland alternatives achieved good values for the added benefits of shoreline access (purple band in Fig. 4) and wetland habitat (dark blue), whereas all other options did not contribute to meeting these two objectives at all. The option of recycling wastewater for irrigation was the only one to fulfill the objective of increasing water supply (light green). For Level 3 upgrades of treatment plants, scoring highly in water quality (light blue), permitting (yellow), and reliability (brown) did not compensate for low scores for the CO_2 emissions (orange) and ease of adaptation (red) objectives and a lack of co-benefits like water supply or wetland habitat for SH4.

3.5. Comparison of MCDA with cost efficiency

Several stakeholders said they would normally assess the value of a nutrient management option through a 'cost-efficiency' measure (mass of nutrients removed/dollar). The results for each option (Fig. 5) assumed the mean cost (total net present value over 30-year technology lifespan) in the uncertainty distributions and the total nitrogen removal (kg of total N removed over a 30-year technology lifespan) in the Status quo scenario. A higher cost-efficiency shown in the figure signifies more nitrogen removed per dollar spent.

Open water wetlands (light green) and optimization (light orange) performed well in the cost-efficiency metric. Wetland levees and recycling for irrigation (which ranked highly for many stakeholders in the MCDA), scored relatively low in the cost-efficiency metric.

Although many stakeholders mentioned cost-efficiency as the "standard" method for decision-making (i.e., the institutionally-sanctioned method) this was not reflected in the MCDA results for many stakeholders. We therefore analyzed the correlation between cost-efficiency and MCDA overall values for each option and stakeholder.

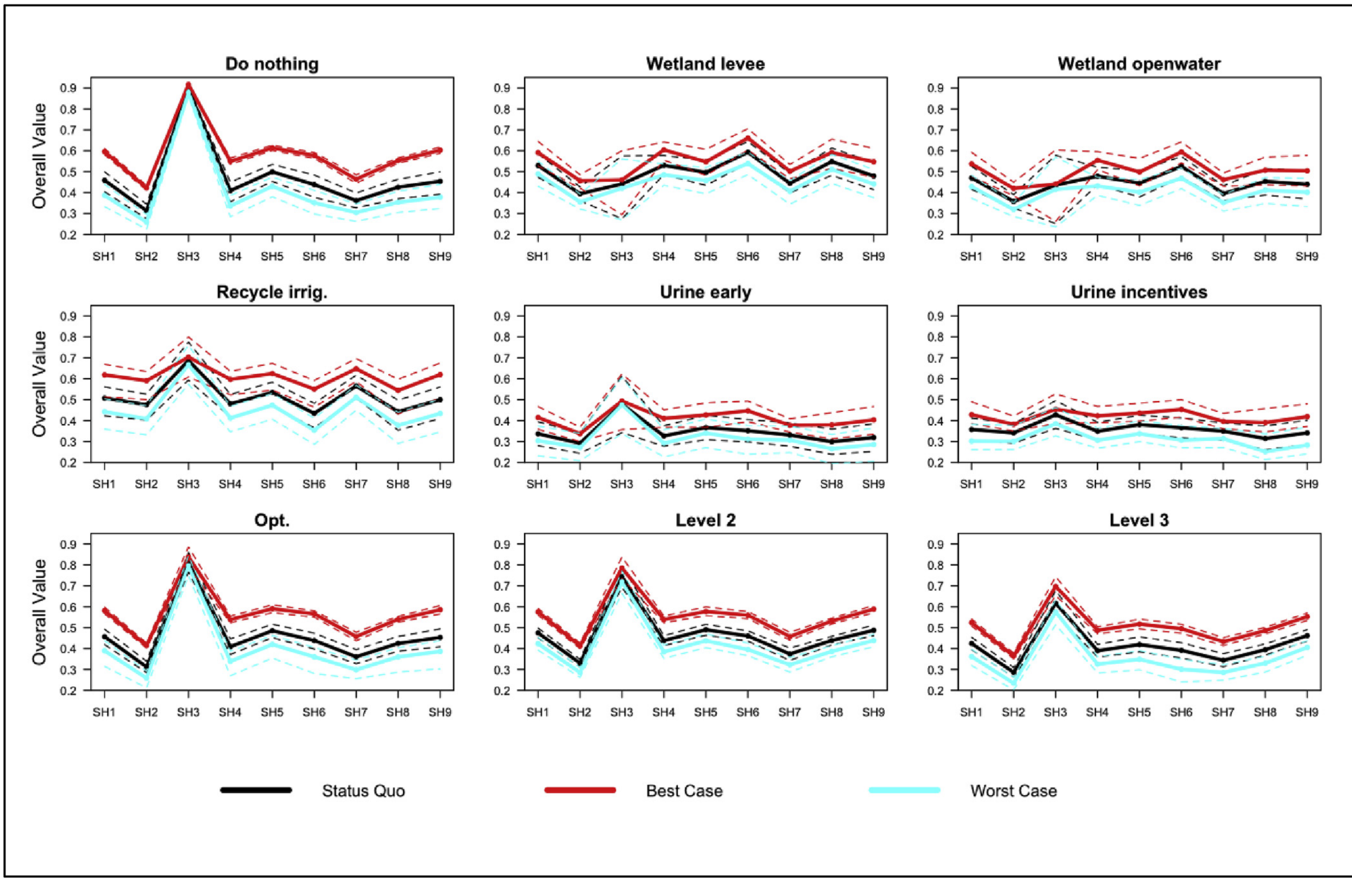

Fig. 2. Overall values for each management option (see Table 1) for each of nine stakeholders (SH; on x-axis) under three future scenarios (lines: status quo, best, and worst case). Solid lines denote median overall values in each scenario, dashed lines represent 5% and 95% quartile values.

Stakeholders exhibited a range in correlations between these two measures (Supplemental Information, Fig. S11). Some stakeholders' preferences for non-traditional goals for nutrient management (which were captured in the MCDA), resulted in a negative correlation between cost-efficiency and MCDA results. Other stakeholders showed a positive correlation between cost-efficiency and MCDA overall value, signifying they valued cost-efficient options in the MCDA.

3.6. Sensitivity analysis

3.6.1. Thresholds

Some stakeholders voiced acceptance thresholds (veto conditions) for particular attributes, below which an option would be unacceptable. Four stakeholders set thresholds in attribute levels for reliability, water quality, ease of adaptation, and resilience to sea level rise. Inclusion of these thresholds in MCDA calculations would change the overall value of options for these stakeholders and, in one case, make all options unacceptable. Complete results of threshold analysis for all stakeholders with stated thresholds are in the Supplemental Information, Text S6.

3.6.2. Aggregation functions

We tested a range of aggregation variants combining additive aggregation with Cobb-Douglas aggregation (Supplemental Information, Fig. S12) (as in Zheng et al., 2016). Cobb-Douglas aggregation tends to value options more highly that do not have extreme variation in levels of attribute fulfillment between objectives. Overall option ranking remained similar regardless of the aggregation type. However, aggregation variants with higher levels of Cobb-Douglas aggregation tended to result in lower overall value for traditional upgrades (which have high fulfillment of some objectives like reliability and no fulfillment of objectives like shoreline access or wetland habitat).

3.6.3. Marginal value functions

We tested the assumption of linear value functions with a comparison of overall value for each option given different shapes of value functions for the objectives ease of adaptation, permitting, reliability, and water quality. These objectives were chosen because several stakeholders expressed mild-to-moderate concave curvatures for them in interviews. We tested a range of curvatures for these four objectives to assess how they affected the overall value of options (Supplemental Information, Fig. S13). Convex value functions imply greatest marginal overall value gained with improvement at low levels of attribute value. Concave value functions imply greatest marginal overall value gained with improvement at high levels of attribute value. Mild-to-moderate concave value functions (what stakeholders expressed in interviews) had little effect on overall ranking of options.

3.6.4. Weight of the objective “total cost”

Stakeholders may have weighted objectives in interviews in ways that did not reflect true weights in decision-making. In particular, many stakeholders minimized the importance of costs in comparison to other objectives. While this may have reflected their feelings, it could also have been a result of answering in ways they thought the researcher would appreciate (i.e., social desirability bias) (Nederhof, 1985). MCDA researchers have noted interview participants tend to weigh objectives at approximately 1/n, where ‘n’ is the number of objectives (Marttunen et al., 2018). This bias holds true for the ‘low cost objective’, where the median cost weight was 0.073, which is approximately equal to 1/13 (0.077).

Given strong institutional mandates for minimizing costs for both

Journal of Environmental Management 233 (2019) 218–237

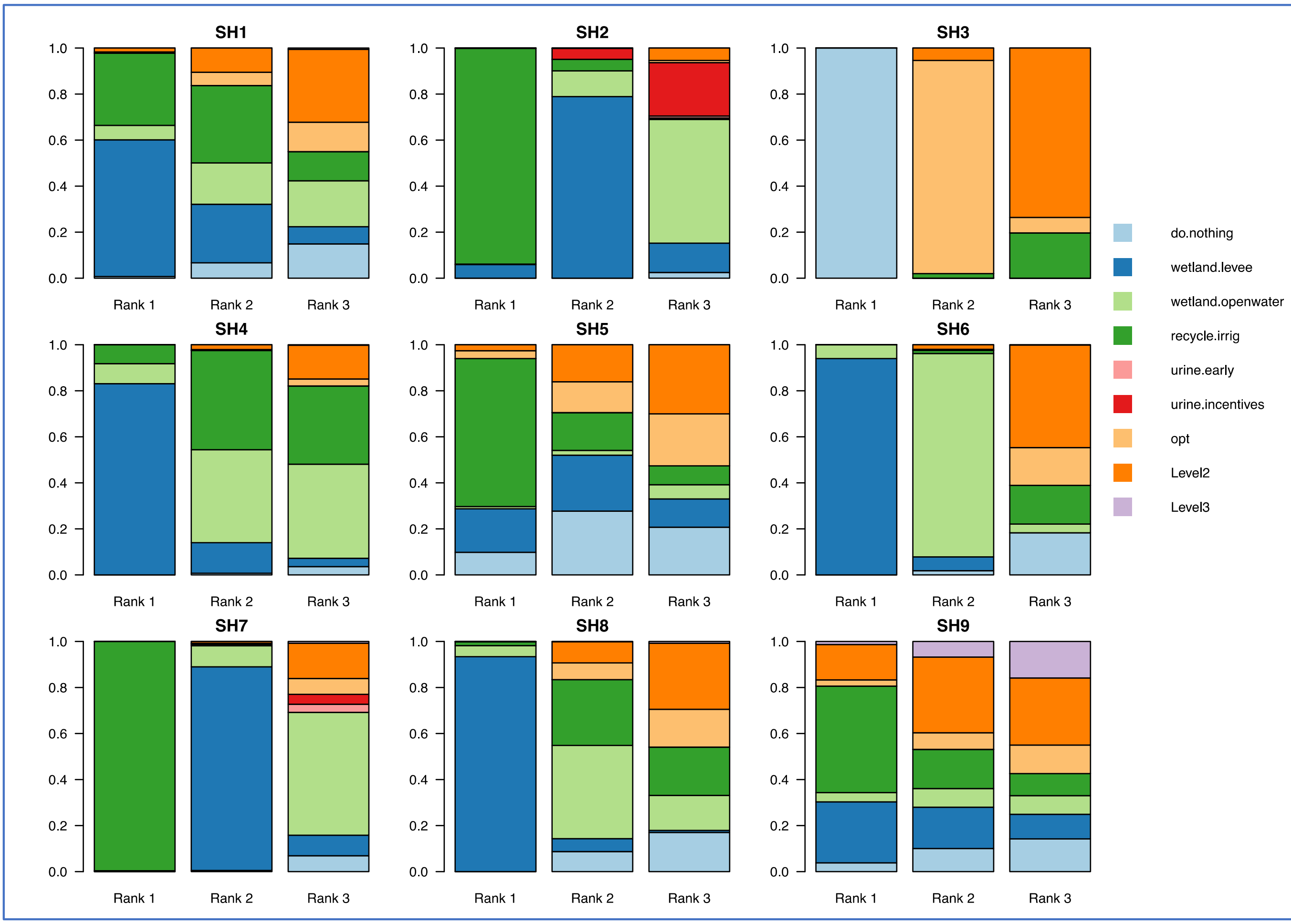

Fig. 3. The probability of the top three ranked options for each of nine stakeholders (SH) given uncertainty in attribute predictions, Status quo scenario. Color coding options see legend and Table 1. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

dischargers and regulators (Harris-Lovett et al., 2018), we increased weights for the objective category "total costs" by 25% (and re-scaled other weights accordingly) to see how these changes influenced overall values of different options in the Status quo scenario (Supplemental Information, Fig. S14). This resulted in an increase in the median overall value of the "Do nothing" option for many stakeholders, though the ranking of the top option for most stakeholders did not change.

4. Discussion

4.1. Implications of MCDA results for nutrient management in the San Francisco Bay case study

The results of the MCDA provide several interesting insights. First, increasing wastewater recycling for irrigation ranks among the top three options for many stakeholders regardless of future conditions (Fig. 3; Supplemental Information, Figs. S6–S10), because it increases water supply and utilizes nutrients in the waste stream for fertilizer (as exemplified for one stakeholder in Fig. 4). This option remained attractive to many stakeholders despite lower cost-efficiency (Fig. 5). Though recycling for irrigation can be expensive – and in some cases has been considered prohibitively so (Bischel et al., 2012) – it may be seen as a viable option if it also prevents nitrogen discharge to sensitive water bodies, because stakeholders' considerations of value go beyond the monetary costs.

Regarding nutrient management, recycling wastewater for irrigation is far superior to recycling wastewater for potable reuse, because nutrients are not removed from irrigation water prior to reuse. Potable water reuse requires safe disposal of concentrate generated during reverse osmosis treatment. Diverting this concentrate away from sensitive water bodies like the Bay is a significant barrier due to costs and greenhouse gas emissions. Treatment technologies to remove nutrients from reverse osmosis concentrate are under development, but most have not been proven in full-scale systems (Umar et al., 2015; Pérez-González et al., 2012). If nutrients could be effectively removed from brine, potable water reuse could become a more feasible option for nutrient management.

Construction of treatment wetlands (horizontal levees) for nutrient treatment also ranked highly for many stakeholders in various uncertain futures (Fig. 3; S6–S10). Additional wetland habitat, increased resilience to sea level rise, shoreline access, and treatment of contaminants of emerging concern favored this option for many stakeholders in the MCDA despite low cost-efficiency (Fig. 5). Because this is a relatively new approach, the cost may be reduced as designers gain more experience with system construction and operation. Furthermore, additional experience will decrease uncertainty about system performance.

A lack of familiarity with urine source-separation technology likely explains its low ranking by most stakeholders. In contrast to the United States, this technology has gained more credibility in Switzerland and Scandinavia (Lienert and Larsen, 2009). If urine source-separation did not require financial incentives to encourage adoption, it would likely be a cost-effective way to avoid nitrogen discharge to the Bay. Additionally, it provides the added benefits of recovering nutrients from

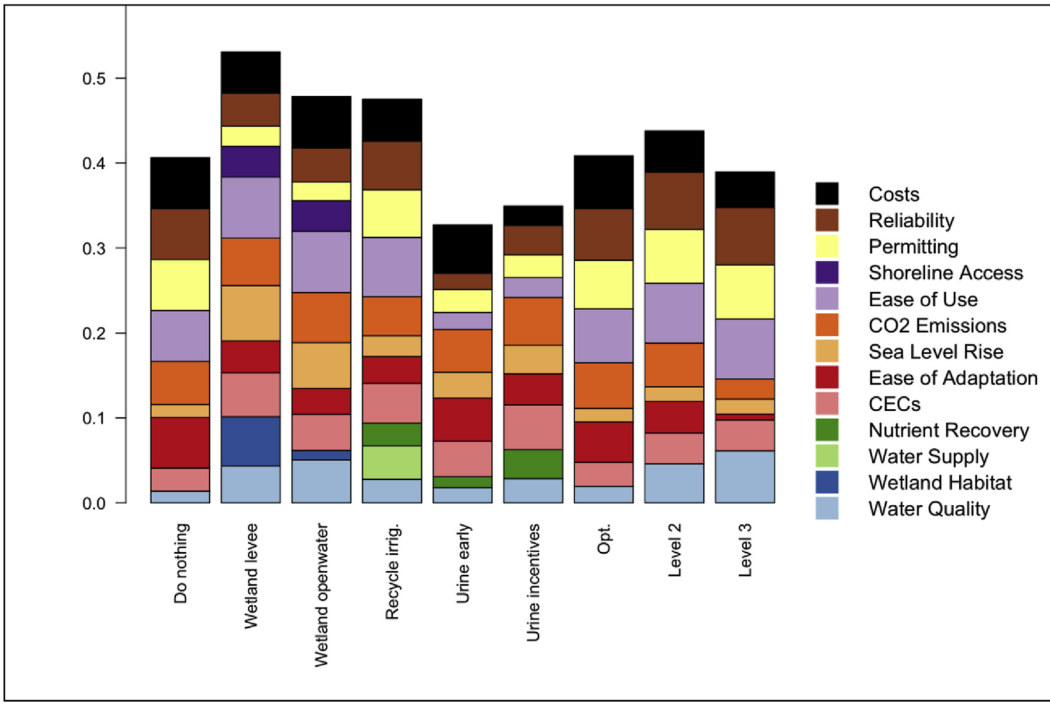

Fig. 4. Median overall value (y-axis) of each option (x-axis; see Table 1), broken down by objectives (in color; see Table 4) for stakeholder SH4 in the Status quo scenario. This figure illustrates for one stakeholder, how strongly each objective contributes to the overall value for each option. Narrow bars: low contribution to overall value; broad bars: large contribution. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

sewage for fertilizer and of being more easily adaptable to changing conditions because it is easily installed in more households if needed. To advance this potentially attractive option, pilot projects to increase public awareness and identify conditions affecting its performance in the United States would be helpful.

Upgrades to existing treatment plants would likely be deployed only in response to regulations related to nutrient loading, because the main benefit of these options is nutrient control. In contrast, multi-benefit options, like increased recycling of wastewater for irrigation and construction of horizontal levees, ranked more highly than upgrades or the

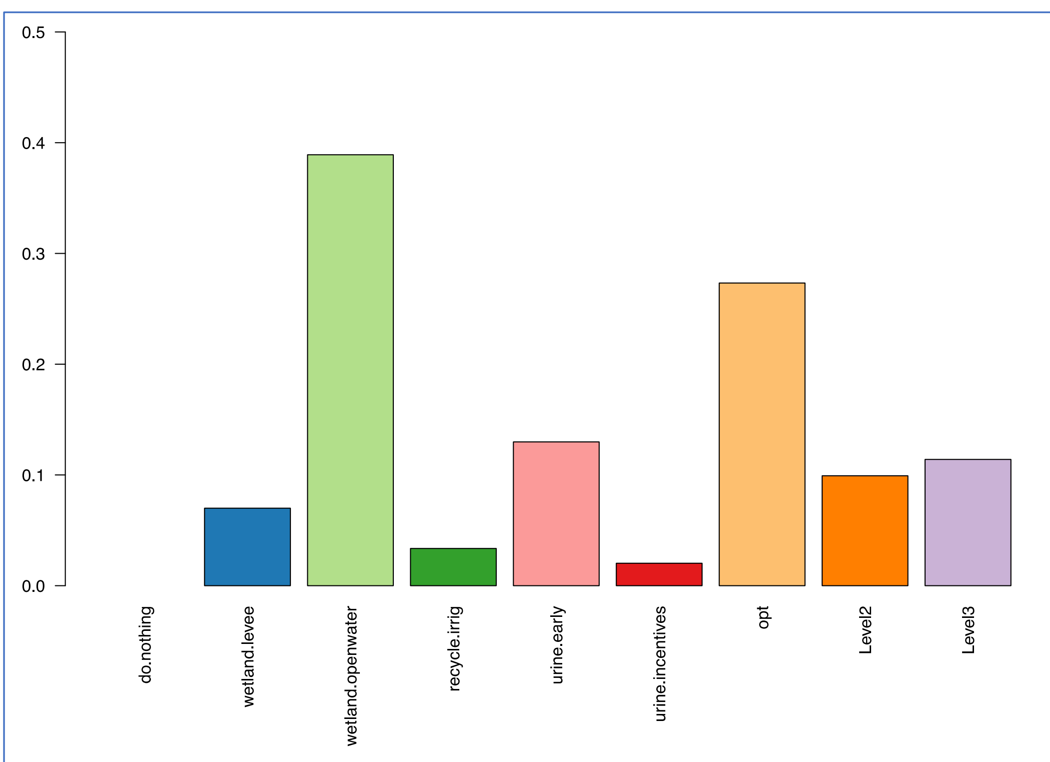

Fig. 5. Mean cost-efficiency (kg of nitrogen removal/$) for each option, Status quo scenario.

'Do nothing' option for most stakeholders even under the Best-case scenario, where there is decreased pressure for nutrient loading to cause adverse ecological effects (Fig. S7).

The overall value for many of the nutrient management options is relatively low (scoring under 0.5) for many stakeholders in the Status quo and Worst-case scenarios. This indicates that in the absence of the Best-case scenario, no single option meets stakeholder goals. Importantly, these options are not mutually exclusive – for example, optimization and constructed wetlands could both be implemented at the same wastewater treatment plant resulting in much lower concentrations of nutrients being released to the Bay. Combinations could provide effective ways to meet more objectives under less-desirable future scenarios.

4.2. Sensitivity analysis

Acceptability thresholds in criteria mentioned by several stakeholders would influence the ranking of options and would remove many options from consideration in any future scenario (see similar observation and discussion in Scholten et al., 2015). This suggests that further conversation about thresholds could determine how stakeholders would react to an exceeded threshold (e.g., litigation, disapproval of project, removal of funding). Efforts to limit scientific uncertainty should focus on areas where stakeholders have stated thresholds (e.g., likelihood of deviation from good water quality, resilience to sea level rise, reliability, and ease of adaptation to changing conditions) to provide better estimates of how well different management options perform for these objectives. Additionally, options that do not exceed stakeholders' stated thresholds should be developed. Technically, such acceptability thresholds can be modeled by minimum aggregation (e.g., Langhans et al., 2014). However, for the purposes of evaluating options for regional environmental management as in the case study, assumed additive aggregation likely provides sufficient insight into management options because rankings were largely unchanged for most stakeholders.

The results of a MCDA (the total value and ranking of options) can be highly sensitive to stakeholders' weights, which is why elicitation of this preference parameter is often critical. In this case study, the weights for "total costs" were lower than researchers expected (median of 0.07; Harris-Lovett et al., 2018). This is in line with a recent meta-analysis concerning weight elicitation procedures in environmental MCDA cases, where economic objectives usually received lower weights than environmental and social objectives (Marttunen et al., 2018). Because interviewees might not fully express institutional economic constraints, sensitivity analysis for this parameter is worth considering. Increasing the weight for "total cost" by 25% increased the overall value and rank of the "Do nothing" option for many stakeholders. This finding is significant because it encourages reflection by decision-makers and policy-makers. Specifically, stakeholders must consider the likelihood of convincing their institutions to overcome a traditional focus on low-cost solutions in order to pursue other goals, which may fall outside of their mandated responsibility (e.g., a wastewater utility funding shoreline access or increased water supply). Further alignment of decision-makers' institutional mandates with their goals for multi-benefit infrastructure and environmental management would reduce uncertainty about stakeholders' ability to follow through on their stated priorities.

4.3. MCDA process integrated with stakeholder analysis and scenario planning

Many useful insights to decision-making about environmental management can be gained from integrating MCDA with stakeholder analysis and scenario planning. This combination of methods sheds light on the ways in which different stakeholders who must collaborate on regional infrastructure projects would value various management options. The addition of scenario planning to MCDA elucidates how management options fare in different possible futures. These insights can help avoid costly investments in infrastructure that may be more controversial or less resilient to future conditions. Many stakeholders in this case have expressed interest in the results. This mixed-methods approach is a promising decision-support tool for other problems in regions that require large-scale collaboration with multiple stakeholders for long-term infrastructure planning and development.

R software was especially useful for combining MCDA with scenario planning, since it allows for easily shifting baseline assumptions about the performance of various management options in the MCDA calculations. Additionally, open-source software like R makes the calculations more transparent and enables stakeholders to vary the future scenario conditions (with more or less population growth, for example).

However, the mixed-methods approach has several limitations. Policy-makers charged with environmental management tend to be averse to complex assessments like these, rather tending to select options that are economically efficient and have low risks (Starkl et al., 2009). Furthermore, time-intensive, in-person interviews with stakeholders may not always be possible (Marttunen and Hämäläinen, 2008). Analysis of qualitative interview data in conjunction with quantitative MCDA requires support of analysts who are versed in multiple methods of inquiry. Finally, without developing a definite 'answer' to the challenge of environmental management it may be unclear to stakeholders how to use the MCDA results to reach consensus in practice – especially if the research protocol anonymizes the contributions of different stakeholders and thus de-personalizes the results.

A summary of the methods employed in this study, along with their advantages and disadvantages, are in Table 5.

4.3.1. Choice of stakeholders

When MCDA is applied to a complex problem involving regional environmental management, it is difficult to choose suitable stakeholders. Because stakeholder preferences affect the analysis, this selection process is crucial. This problem is common in decision-making or strategic planning situations in which there is a desire for stakeholder engagement. However, it remains a salient issue, though best practices include efforts to include those affected by decisions as well as decision-makers and expanding beyond the 'usual suspects' (see for example, Colvin et al., 2016; Reed et al., 2009; Achterkamp and Vos, 2007; Vos and Achterkamp, 2006; Bryson, 2004; Mitchell et al., 1997). Note that stakeholder choice will vary over time, as stakeholders who are currently most important to decision-making may change (Brugha and Varvasovszky, 2000). This was evident even in this case study. Several people who participated in the first set of interviews changed jobs by the time the follow-up interviews occurred several months later.

Our method of stakeholder selection emphasized diversity of opinion and profession. Despite our efforts to obtain a broad range of opinions, many of our chosen stakeholders were technical experts or represented government/municipal agencies. This tendency to emphasize expert opinions has been observed in previous MCDA studies (Soltani et al., 2015). For example, several groups that might have expressed different perspectives were not included in the interviews (e.g., subsistence fishers, farmers who might use recycled water for irrigation) because they engaged less publicly with the issue.

In addition, after an organization was positively identified as a "stakeholder", the most appropriate person to interview was not always evident (e.g., managers versus board members). Other researchers have also noted that transparent methods to select and engage stakeholders in a participatory MCDA process are not available (Marttunen et al., 2015). Further research to improve the stakeholder identification process, especially for application to regional environmental MCDA, would be a useful contribution to the field.

Clustering and selecting participants for follow-up interviews based on their goals for nutrient management and their involvement with the issue clarified the differences in objectives amongst people with similar

Table 5
Methods integrated into the standard MCDA process to support regional decision-making for multi-benefit infrastructure.

Step	Methods	Advantage	Disadvantage
Initial stakeholder selection	Broad outreach to people and organizations who have authored documents or participated in public meetings related to nutrient management	- Includes perspectives of those who have been publicly working on the issue	- Not necessarily clear whom to include within an organization - Stakeholders with less time or influence, who may still have strong feelings about the issue, are not included
	Snowball sampling	- Personal referrals to targeted individuals - Gain insight into identity of influential stakeholders	- Can lead to sampling within a 'bubble' of people with similar ideas or professional roles, might neglect important stakeholders from other professional fields or regions
	Inclusion of stakeholders whom researchers deemed would be affected by nutrient management but who were not involved in authoring documents, public meetings, or recommended by other interviewees	- Can include perspectives of marginalized groups who have not traditionally been included in decision-making about environmental management	- Researchers may not be able to accurately predict who would be affected
Selection of stakeholders for MCDA preference elicitation interviews	Cluster analysis based on stated goals for nutrient management in initial interviews, followed by stratified sampling to choose those stakeholders most relevant to decision-making from each cluster	- Broad representation of stakeholders with different goals - Does not assume stakeholders with same professional role necessarily have the same goals	- May over- or under-represent stakeholders from any particular professional role
Identify objectives and attributes	Solicitation of objectives and attributes from individual stakeholders (rather than focus groups or stakeholder workshops)	- Encourages participation from stakeholders with less influence or political power - Identifies areas of disagreement and agreement among stakeholders	- Differences in language among different stakeholders may result in researcher misinterpretation of objectives and attributes - No consensus reached on objectives and attributes
	Researcher synthesis of objectives and attributes	- Encourages the inclusion of objectives from stakeholders with less influence or political power	- May result in disagreement about the accuracy of attributes for describing objectives - Choices about structuring the objectives hierarchy can bias stakeholder weights
	Limited objectives to < 15	- Ease of mental processing for stakeholders - Less time required for elicitation	- Possible consolidation of objectives that some stakeholders consider distinct
	Researchers generated objectives that helped differentiate between management options	- Assisted with differentiation between specific management options in the MCDA	- May be less relevant to stakeholders than some of the other objectives
Development of future scenarios	Informed by stakeholder ideas about "critical uncertainties" that would affect nutrient management	- Takes into account the uncertainties stakeholders are considering	- Does not take into account unforeseen situations that could strongly affect future conditions
	Used scenario generation matrix to develop Best- and Worst-case scenarios	- Bounds uncertain futures within the areas specified	- Best- and Worst-case scenarios may be less useful for prescriptive MCDA when choosing management options
Development of management options	Illustrative options applied at their maximum extent to the whole region, rather than more realistic combinations of options or site-specific options within the region	- Highlights ways in which different management options can fulfill different stakeholder objectives	- Is not realistic, does not provide a prescriptive set of actions
Predict outcomes of each option, given uncertainty about the future	Estimated range of values and distribution for attributes from the literature, from expert opinion, and modeled from previous technology implementation	- Approximates uncertainty in attribute values for all objectives - Elucidates magnitude of differences in MCDA results due to uncertainty in technical attribute prediction versus future scenario conditions	- Distribution of uncertainty could be incorrect - Attribute values from the literature and past implementation may be quite different from local values due to local conditions
Elicitation of stakeholder preferences (weights and marginal value functions)	Used notecards of objectives stakeholders could physically move around on the table to represent preference weights for Swing method elicitation	- Allows for kinetic experience of the weightings - Stakeholders can easily re-arrange to 'try' different weights and see what seems most accurate	- Requires in-person interviews - Time intensive
	Did not include stakeholder uncertainty in preference weights or consideration of differences in weights from a personal vs. professional perspective	- Simplifies MCDA - Less time-intensive	- May inaccurately represent stakeholder preferences
	Used bi-section method to elicit marginal value functions	- Has the potential to provide rough curvature estimates with little elicitation time	- May result in vague, non-quantitative results due to political nature of making some values explicit or time constraints
Integrate preference weights and attribute predictions to rank options	Performed rankings for all stakeholders separately, did not use aggregate by using average weights	- Identifies the range of stakeholder opinions - Identifies areas of conflict and agreement amongst stakeholders	- Complex to analyze and interpret results - No clear 'answer' from the MCDA regarding a consensus solution; this would require further stakeholder workshops

professional roles. Because we deliberately chose stakeholders who reflected a range of opinions, some participants expressed opinions that may be considered as outliers. The importance of this sampling method was validated in interviews, where several participants expressed the opinion that individuals with strong opinions could have an outsize effect on the decision-making process through litigation or soliciting media attention.

The cluster analysis informed the selection of participants for second-round interviews who had a range of viewpoints and who were most relevant to decision making. The fact that all stakeholders who were asked to participate in follow-up interviews agreed to do so, despite the significant time investment (see Section 4.3.4), confirms the relevance of the clustering methodology for stakeholder selection.

4.3.2. Identification of objectives and attributes

Identifying objectives and attributes based on interviews yielded essential information, but the process was not always orderly. Similarly worded objectives could have different meanings to participants. As a result, we had to make subjective decisions to condense the plethora of stated objectives into a manageable set (Harris-Lovett et al., 2018). Additionally, we made subjective choices to categorize objectives (e.g., 'ease of permitting' could have fit into the categories of 'low costs' or 'social support'). These choices could affect weighting of the objectives, depending on how many other objectives were in the category and whether they emphasized social or economic values (Marttunen et al., 2018). Providing participants with a list of potential objectives and allowing them to place them into categories (after they came up with their own objectives) might allow researchers to standardize differences in understanding of language among stakeholders.

It was also difficult to identify measureable attributes for each objective. Others recommend stakeholder workshops to develop such attributes (Eisenführ et al., 2010; Belton and Pictet, 1997; Massey and Wallace, 1996). However, such a time-consuming process might not be feasible or agreeable to all participants. As an alternative, we developed our objectives based on ideas expressed by individual stakeholders during the interviews; this allowed individuals with less decision-making influence to have their opinions incorporated into the study design.

Despite these challenges, defining the objectives hierarchy and selecting measurable attributes were some of the most instructive steps of the MCDA process. Vagueness of stated objectives in interviews and discrepancies among opinions about proper measurement attributes are areas where further research and discussion amongst stakeholders may be beneficial. In this case study, these included a need for developing technologies for nutrient management that are easily adaptable to changing conditions and clearly defining criteria for nutrient-related impairment to the San Francisco Bay ecosystem.

4.3.3. Choice of management options

Generating management options is an integral part of the "problem structuring" aspect of MCDA (Belton and Stewart, 2002). Other MCDA analysts suggest choosing options that emphasize fulfillment of different objectives (Pereira et al., 1994). In environmental decision-making, researchers have emphasized the importance of ensuring stakeholder participation and including both standard and innovative options (Lahdelma et al., 2000). However, little guidance exists for determining the scale of the options that need to be considered in regional environmental decisions. In this case study, it was not clear whether the MCDA problem should be considered at the scale of a single wastewater treatment plant (much simpler to analyze) or for the whole southern reach of the Bay (complex but more relevant to actual decision-making). At an early point in the environmental decision-making process about a regional challenge (before any regulations have been set, in this case about nutrient loading), and with so many quasi-independent decision-makers, it is difficult to develop options to model with MCDA that accurately represent actual management options.

To simplify this problem, we applied each option across wastewater treatment plant service areas. This highlights the general benefits, drawbacks, and discussion points of each management option, but does not provide personalized guidance to managers about the specific options that would be optimal for their situation. Further research could assess whether this approach yields results that are substantially different from MCDAs that consider multiple smaller-scale options for different areas within a larger region. Comparison of MCDA results using modeled regional management options to local ones would be one way of conducting such further research.

4.3.4. Elicitation of stakeholder preferences

To evaluate the reliability of stakeholders' weights for objectives derived from the Swing elicitation process, they were asked: "How much did you take into account the worst and best values of each goal when you decided on the swings?" (Harris-Lovett et al., 2018). Seven stakeholders answered (two did not because of time constraints), but only one chose the response option: "They were essential to my decision." Four responded: "I took them into consideration", and two: "I didn't consider them". Range insensitivity is well-known in the MCDA literature (Clemen and Reilly, 2004). To assure accurate results, weight elicitation methods rely upon respondents' careful deliberation of the worst-best attribute range (Eisenführ et al., 2010). Although this requirement might have been violated to some extent in our case, the elicitation process may still have been useful for decision-makers (Marttunen and Hämäläinen, 2008). People often form their preferences in the process of assigning weights (Belton and Stewart, 2002), so weight elicitation helps decision-makers clarify their beliefs.

It is good practice to carry out consistency checks of elicited weights with another method; the trade-off method can serve as an alternative to the Swing method (Eisenführ et al., 2010). Despite its strong theoretical foundation (Keeney and Raiffa, 1976) and its usefulness in defining stakeholder preferences in MCDA for environmental management (Reichert et al., 2015), we found the trade-off weight elicitation method to be ineffective in practice. All stakeholders were unwilling to express numerical trade-offs between attributes (e.g., "Paying $200,000 to reduce the risk of impairing water quality by 5% is better than paying $600,0000 to reduce the risk of impairing water quality by 20%"). These types of value statements are cognitively difficult, as well as highly political. Explicit identification of trade-offs may heighten conflict and lower confidence in decision-making (Kottemann and Davis, 1991). This could explain the reluctance of decision-makers to accept this premise in the interview process. Although some MCDA analysts have successfully employed the trade-off method in similar contexts (Anderson et al., 2001), our experience was consistent with that of researchers who found many stakeholders reluctant to express numeric values for trade-offs (Zheng et al., 2016).

Similarly, we found marginal value functions difficult to elicit with the bisection method ("Improvement from the worst value to a middle-value point is perceived as equally beneficial as improvement from a middle-value point to the best value"). These questions were almost uniformly met with vague, non-quantitative responses, possibly due to time constraints or an inability to give numeric values for complex decisions. Some of the difficulty may have been due to the relative inexperience of the researcher conducting MCDA interviews (Pöyhönen and Hämäläinen, 2000). The sensitivity analysis indicates that differences in marginal value function curvatures (particularly for convex value functions) could drastically change the overall value of options for many stakeholders. Therefore, avoiding the difficulties by assuming linearity of value functions would be inadequate for this case (e.g. Langhans and Lienert, 2016; Zheng et al., 2016). Thus, other methods of eliciting marginal value functions would have been needed to accomplish these goals.

We found that allocating one hour for follow-up interviews for preference elicitation was not always sufficient. This poses a problem for MCDA interviews because many high-level decision-makers have busy schedules; an observation shared by many researchers implementing MCDA. We recommend that future research should develop tools that allow for reliable preference elicitation in a more efficient manner. One idea receiving increasing interest for reducing interview length is adaptive elicitation, where decision-makers' specific answers determine the questions that are asked next (Ciomek et al., 2017; de Almeida et al., 2016).

Our experiences suggest that the main insights gained from the elicitation of stakeholder preferences are qualitative, and are obtained from conversation about the value of different objectives. For example, attempts to elicit bi-section values for the objective 'reliability' led to reflections about reliability for wastewater management in general, compared to nutrient management in particular. Managers desire reliability close to 100% for controlling pathogens, because any lapse can potentially affect public health. In contrast, occasional periods of high nutrient concentrations in effluent due to lapses in reliability are not likely to be ecologically detrimental if their duration is limited.

Interviews also revealed tensions between personal values, professional roles, and institutional mandates. For example, several stakeholders mentioned that they personally valued wetland habitat highly but that this was not within the scope of their professional role. To obtain results that were most reflective of the actual decision-making process, we asked stakeholders to weight criteria based on whatever value (personal or professional) would affect their professional decision-making. However, this made it difficult for several stakeholders to answer the questions. Future MCDA procedures could include elicitation of value judgements both as a private person and as an official representative, with sensitivity analysis to compare any differences in results. Additionally, improved understanding of how private values influence professional actions in regional environmental decision-making would provide insight into the importance of this potential conflict.

4.3.5. Addressing uncertainty of stakeholder preferences, future scenarios, and attribute predictions

We opted to keep stakeholder weights separate, rather than aggregate them. This allowed us to highlight the diversity of opinions and to assess management options ranked highly by multiple stakeholders (Gregory et al., 2001; Matsatsinis and Samaras, 2001; Belton and Pictet, 1997). However, disaggregated results do not provide decision-makers with a clear path forward. Instead, they raise questions about the importance of different stakeholders' opinions. The MCDA research protocol precludes identification of the stakeholders, so disaggregated group results only provide information about topics of disagreement that may require attention, not about individuals who disagree. In situations where disparate opinions are observed, follow-up stakeholder workshops could be used to promote discussion and build consensus (Ferguson et al., 2013).

Future scenarios were selected to illustrate the robustness of MCDA options in the year 2050 (Marttunen et al., 2017). By designing future scenarios with the “critical uncertainties” stakeholders mentioned, we aimed to capture the range of future possibilities that would likely have the biggest impact on nutrient management. However, this approach considers only the “known-unknowns” – and none of the “unknown-unknowns”. “Unknown-unknowns” could originate in another sector entirely (e.g., regulations on greenhouse gas emissions) and could deeply constrain wastewater treatment operations by making a specific technology much less attractive, for example. Characterization of potential “unknown-unknowns” and analysis of the MCDA results under these less-predictable conditions could enhance the reliability of the MCDA.

Including uncertainty in attribute predictions in the MCDA highlighted the ways in which uncertainty about technical performance of the management options could affect outcomes (Zheng et al., 2016; Durbach and Stewart, 2012). This was especially useful in combination with future scenarios because it allowed for differentiation between uncertainty that cannot be controlled (future conditions) and uncertainty that can be minimized by collecting additional data (attribute predictions). In this case study, minimizing uncertainty related to attribute predictions would clarify rankings of outcomes, because several options for stakeholders spanned very different ranks (from 1st to 6th, for example) depending on attribute predictions (Supplemental Information, Figs. S8–S10). Future scenarios, in contrast, largely changed overall values for options rather than their relative ranking (Supplemental Information, Fig. S5).

5. Conclusions

The use of stakeholder analysis, MCDA and scenario planning in the San Francisco Bay case study provided critical insight. Stakeholders with similar professional roles (e.g., dischargers) often held different objectives for “good nutrient management”, which implies that sampling methodologies for gaining stakeholder input should go beyond sampling people in different professional roles. Our results illuminated that increased wastewater recycling for irrigation performed highly as a nutrient management option for most stakeholders regardless of future conditions, because it increased water supply. Likewise, constructed treatment wetlands performed well, because of their multiple benefits like increased wildlife habitat, resilience to sea level rise, and treatment of contaminants of emerging concern. Stakeholders in the case study have expressed interest in the results and have begun to discuss the implications of these findings for decision-making.

This mixed-methods approach to strategic planning in environmental management provides useful support to the decision-making process in our case study, and is likely transferable to other cases as well. Stakeholder analysis and MCDA paired with analysis of future uncertainties can integrate stakeholders' perspectives into formulating goals for environmental decision-making, and assess the ways in which different management options fulfill these goals. These methods can highlight areas of agreement amongst stakeholders, laying the groundwork for discussion and collaboration. They can differentiate between the uncertainties over which decision-makers have little control (e.g., future conditions) and the uncertainties which additional data collection, research and development can minimize (e.g., modeled attribute predictions). Additionally, these methods can incorporate the perspectives of potentially important stakeholders who may have been excluded from traditional decision processes.

Integrating stakeholder analysis, MCDA, and scenario planning can support regional environmental decision-making and merits further research. Our method of applying cluster analysis to select stakeholders for in-depth MCDA interviews, rather than selection based solely on their professional role, could be further refined in other research contexts. Explicit consideration of who has been included, and why, in group decision analyses for environmental management could illuminate other methods for selecting stakeholders to participate in different stages of the MCDA and clarify for researchers the most appropriate method(s) for their own research. This type of review could support efforts to make environmental decision-making more equitable. Finally, testing and refining the combination of stakeholder analysis, MCDA, and scenario planning in other environmental management contexts would be a fruitful area for further inquiry.

Declarations of interest

None.

Acknowledgements

The authors would like to thank all the interview participants for their time and thoughtful contributions to this research. We gratefully acknowledge the financial support from the NSF Graduate Research Fellowship Program (SHL). This research was supported by the National Science Foundation (NSF) through the Engineering Research Center for Reinventing the Nation's Water Infrastructure (ReNUWIt) EEC-1028968.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2018.11.112.

References

Achterkamp, M.C., Vos, J.F., 2007. Critically identifying stakeholders. Syst. Res. Behav. Sci. 24, 3–14.

Al-Kloub, B., Al-Shemmeri, T., Pearman, A., 1997. The role of weights in multi-criteria decision aid, and the ranking of water projects in Jordan. Eur. J. Oper. Res. 99, 278–288. https://doi.org/10.1016/S0377-2217(96)00051-3.

Alpine, A.E., Cloern, J.E., 1988. Phytoplankton growth rates in a light-limited environment, San Francisco Bay. Mar. Ecol. Prog. Ser. 167–173.

Anderson, D.M., Glibert, P.M., Burkholder, J.M., 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25, 704–726.

Anderson, R., Hobbs, B., Koonce, J., Locci, A., 2001. Using decision analysis to choose phosphorus targets for Lake Erie. Environ. Manag. 27, 235–252. https://doi.org/10.1007/s002670010146.

Association of Bay Area Governments and the Metropolitan Transportation Commission, 2017. Plan Bay Area 2040 Draft Plan.

Association of Bay Area Governments, 2016. Regional Forecast for Plan Bay Area 2040. Available online at: https://abag.ca.gov/planning/research/memos/Regional_Forecast_for_Plan_Bay_Area_2040_F_030116.pdf, Accessed date: 3 October 2018.

Atkinson, R., Flint, J., 2001. Accessing hidden and hard-to-reach populations: Snowball research strategies. Soc. Res. Update 33, 1–4.

Balint, P.J., Stewart, R.E., Desai, A., 2011. Wicked Environmental Problems: Managing Uncertainty and Conflict. Island Press.

Batt, A.L., Kim, S., Aga, D.S., 2007. Comparison of the occurrence of antibiotics in four full-scale wastewater treatment plants with varying designs and operations. Chemosphere 68, 428–435. https://doi.org/10.1016/j.chemosphere.2007.01.008.

Belton, V., Pictet, J., 1997. A framework for group decision using a MCDA model: sharing, aggregating or comparing individual information? J. Decis. Syst. 6, 283–303. https://doi.org/10.1080/12460125.1997.10511726.

Belton, V., Stewart, T., 2002. Multiple criteria decision analysis: an integrated approach. Springer.

Benda, L.E., Poff, L.N., Tague, C., Palmer, M.A., Pizzuto, J., Cooper, S., Stanley, E., Moglen, G., 2002. How to avoid train wrecks when using science in environmental problem solving. Bioscience 52, 1127–1136. https://doi.org/10.1641/0006-3568(2002)052[1127:HTATWW]2.0.CO;2.

Biernacki, P., Waldorf, D., 1981. Snowball sampling: Problems and techniques of chain referral sampling. Socio. Methods Res. 10, 141–163.

Bischel, H.N., Simon, G.L., Frisby, T.M., Luthy, R.G., 2012. Management experiences and trends for water reuse implementation in Northern California. Environ. Sci. Technol. 46, 180–188. https://doi.org/10.1021/es202725e.

Bojórquez-Tapia, L.A., Sánchez-Colon, S., Florez, A., 2005. Building consensus in environmental impact assessment through multicriteria modeling and sensitivity analysis. Environ. Manag. 36, 469–481. https://doi.org/10.1007/s00267-004-0127-5.

Borcard, D., Gillet, F., Legendre, P., 2011. Numerical Ecology with R, 2011 edition. Springer, New York.

Borsuk, M., Clemen, R., Maguire, L., Reckhow, K., 2001. Stakeholder values and scientific modeling in the Neuse River Watershed. Group Decis. Negot. 10, 355–373. https://doi.org/10.1023/A:1011231801266.

Boynton, W.R., Kemp, W.M., Keefe, C.W., 1982. A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production. In: Estuarine Comparisons. Elsevier, pp. 69–90.

Brown, R., Farrelly, M.A., 2009. Delivering sustainable urban water management: a review of the hurdles we face. Water Sci. Technol. 59, 839–846.

Brugha, R., Varvasovszky, Z., 2000. Stakeholder analysis: a review. Health Pol. Plann. 15, 239–246. https://doi.org/10.1093/heapol/15.3.239.

Bryson, J.M., 2004. What to do when stakeholders matter: stakeholder identification and analysis techniques. Publ. Manag. Rev. 6, 21–53.

Ciomek, K., Kadziński, M., Tervonen, T., 2017. Heuristics for prioritizing pair-wise elicitation questions with additive multi-attribute value models. Omega 71, 27–45.

Clemen, R.T., Reilly, T., 2004. Making Hard Decisions with Decision Tools Suite Update Edition, 1 edition. Cengage Learning, Pacific Grove, CA.

Cloern, J.E., 1999. The relative importance of light and nutrient limitation of phytoplankton growth: a simple index of coastal ecosystem sensitivity to nutrient enrichment. Aquat. Ecol. 33, 3–15.

Cloern, J.E., Jassby, A.D., 2012. Drivers of change in estuarine-coastal ecosystems: discoveries from four decades of study in San Francisco Bay. Rev. Geophys. 50.

Cole, B.E., Cloern, J.E., 1984. Significance of biomass and light availability to phytoplankton productivity in San Francisco Bay. Mar. Ecol. Prog. Ser. 15–24.

Colvin, R.M., Witt, G.B., Lacey, J., 2016. Approaches to identifying stakeholders in environmental management: insights from practitioners to go beyond the 'usual suspects'. Land Use Pol. 52, 266–276.

Corominas, L., Larsen, H., Flores-Alsina, X., Vanrolleghem, P., 2013. Including Life Cycle Assessment for decision-making in controlling wastewater nutrient removal systems. J. Environ. Manag. 128, 759–767. https://doi.org/10.1016/j.jenvman.2013.06.002.

de Almeida, A.T., de Almeida, J.A., Costa, A.P.C.S., de Almeida-Filho, A.T., 2016. A new method for elicitation of criteria weights in additive models: Flexible and interactive tradeoff. Eur. J. Oper. Res. 250, 179–191.

DeFries, R., Nagendra, H., 2017. Ecosystem management as a wicked problem. Science 356, 265–270. https://doi.org/10.1126/science.aal1950.

Delignette-Muller, M.L., Dutang, C., 2015. fitdistrplus: An R package for fitting distributions. J. Stat. Softw. 64, 1–34.

Dodds, W.K., Bouska, W.W., Eitzmann, J.L., Pilger, T.J., Pitts, K.L., Riley, A.J., Schloesser, J.T., Thornbrugh, D.J., 2008. Eutrophication of U.S. Freshwaters: Analysis of Potential Economic Damages. [WWW Document]. https://doi.org/10.1021/es801217q.

Dominguez, D., Gujer, W., 2006. Evolution of a wastewater treatment plant challenges traditional design concepts. Water Res. 40, 1389–1396. https://doi.org/10.1016/j.watres.2006.01.034.

Dominguez, D., Truffer, B., Gujer, W., 2011. Tackling uncertainties in infrastructure sectors through strategic planning: The contribution of discursive approaches in the urban water sector. Water Pol. 13, 299–316.

Dominguez, D., Worch, H., Markard, J., Truffer, B., Gujer, W., 2009. Closing the capability gap: strategic planning for the infrastructure sector. Calif. Manag. Rev. 51, 30–50.

Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr. 67, 345–366.

Durbach, I.N., Stewart, T.J., 2012. Modeling uncertainty in multi-criteria decision analysis. Eur. J. Oper. Res. 223, 1–14. https://doi.org/10.1016/j.ejor.2012.04.038.

Eisenführ, F., Weber, M., Langer, T., 2010. Rational decision making. Springer, Berlin; London.

Ferguson, B.C., Frantzeskaki, N., Brown, R.R., 2013. A strategic program for transitioning to a Water Sensitive City. Landsc. Urban Plann. 117, 32–45. https://doi.org/10.1016/j.landurbplan.2013.04.016.

Fratini, C.F., Elle, M., Jensen, M.B., Mikkelsen, P.S., 2012. A conceptual framework for addressing complexity and unfolding transition dynamics when developing sustainable adaption strategies in urban water management. Water Sci. Technol. 66, 2393–2401. https://doi.org/10.2166/wst.2012.442.

Garrido-Baserba, M., Reif, R., Molinos-Senante, M., Larrea, L., Castillo, A., Verdaguer, M., Poch, M., 2016. Application of a multi-criteria decision model to select of design choices for WWTPs. Clean Technol. Environ. Policy 18, 1097–1109. https://doi.org/10.1007/s10098-016-1099-x.

Greening, L., Bernow, S., 2004. Design of coordinated energy and environmental policies: use of multi-criteria decision-making. Energy Pol. 32, 721–735.

Gregory, R., Failing, L., Higgins, P., 2006. Adaptive management and environmental decision making: A case study application to water use planning. Ecol. Econ. 58, 434–447. https://doi.org/10.1016/j.ecolecon.2005.07.020.

Gregory, R., McDaniels, T., Fields, D., 2001. Decision aiding, not dispute resolution: creating insights through structured environmental decisions. J. Pol. Anal. Manag. 20, 415–432. https://doi.org/10.1002/pam.1001.

Grimble, R., Wellard, K., 1997. Stakeholder methodologies in natural resource management: a review of principles, contexts, experiences and opportunities. Agric. Syst. 55, 173–193.

Hager, S.W., Schemel, L.E., 1992. Sources of nitrogen and phosphorus to Northern San Francisco Bay. Estuaries 15, 40–52. https://doi.org/10.2307/1352708.

Hajkowicz, S., Collins, K., 2006. A review of multiple criteria analysis for water resource planning and management. Water Resour. Manag. 21, 1553–1566. https://doi.org/10.1007/s11269-006-9112-5.

Hajkowicz, S.A., 2008. Supporting multi-stakeholder environmental decisions. J. Environ. Manag. 88, 607–614. https://doi.org/10.1016/j.jenvman.2007.03.020.

Hämäläinen, R.P., Alaja, S., 2008. The threat of weighting biases in environmental decision analysis. Ecol. Econ. 68, 556–569.

Harris-Lovett, S., Lienert, J., Sedlak, S., 2018. Towards a new paradigm of urban water infrastructure: identifying goals and strategies to support multi-benefit municipal wastewater treatment. Water 10 (9), 1127.

Harris-Lovett, S., Sedlak, D., 2015. The history of water reuse in California. In: Lassiter, Allison (Ed.), Sustainable Water: Challenges and Solutions from California. University of California Press, Berkeley.

Heberger, M., Cooley, H., Herrera, P., Gleick, P.H., Moore, E., 2009. The impacts of sea-level rise on the California coast. prepared for the California Climate Change Center. .

Heisler, J., Glibert, P., Burkholder, J., Anderson, D., Cochlan, W., Dennison, W., Gobler, C., Dortch, Q., Heil, C., Humphries, E., Lewitus, A., Magnien, R., Marshall, H., Sellner, K., Stockwell, D., Stoecker, D., Suddleson, M., 2008. Eutrophication and harmful algal blooms: a scientific consensus. Harmful Algae 8, 3–13. https://doi.org/10.1016/j.hal.2008.08.006.

Hermans, C., Erickson, J., Noordewier, T., Sheldon, A., Kline, M., 2007. Collaborative environmental planning in river management: An application of multicriteria decision analysis in the White River Watershed in Vermont. J. Environ. Manag. 84, 534–546. https://doi.org/10.1016/j.jenvman.2006.07.013.

Howarth, R.W., 2008. Coastal nitrogen pollution: a review of sources and trends globally and regionally. Harmful Algae 8, 14–20.

Howarth, R.W., Anderson, D.B., Cloern, J.E., Elfring, C., Hopkinson, C.S., Lapointe, B., Maloney, T.J., Marcus, N., McGlathery, K., Sharpley, A.N., 2000. Issues in ecology: Nutrient pollution of coastal rivers, bays, and seas.

Howarth, R.W., Sharpley, A., Walker, D., 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. Estuaries 25, 656–676.

Huang, I.B., Keisler, J., Linkov, I., 2011. Multi-criteria decision analysis in environmental sciences: Ten years of applications and trends. Sci. Total Environ. 409, 3578–3594. https://doi.org/10.1016/j.scitotenv.2011.06.022.

Jasper, J.T., Jones, Z.L., Sharp, J.O., Sedlak, D.L., 2014. Biotransformation of trace organic contaminants in open-water unit process treatment wetlands. Environ. Sci. Technol. 48, 5136–5144. https://doi.org/10.1021/es500351e.

Jasper, J.T., Sedlak, D.L., 2013. Phototransformation of wastewater-derived trace organic contaminants in open-water unit process treatment wetlands. Environ. Sci. Technol. 47, 10781–10790. https://doi.org/10.1021/es304334w.

Kabir, G., Sadiq, R., Tesfamariam, S., 2014. A review of multi-criteria decision-making methods for infrastructure management. Struct. Infrastruct. Eng. 10, 1176–1210. https://doi.org/10.1080/15732479.2013.795978.

Kang, D., Lansey, K., 2012. Multiperiod planning of water supply infrastructure based on

scenario analysis. J. Water Resour. Plann. Manag. 140, 40–54.
Kangas, Kangas, A., Pekka, L., Pykäläinen, J., 2002. MCDM methods in strategic planning of forestry on state-owned lands in Finland: applications and experiences. J. Multi-Criteria Decis. Anal. 10, 257–271. https://doi.org/10.1002/mcda.306.
Keeney, R., 1992. Value-focused thinking: a path to creative decision making. Harvard University Press, Cambridge, MA.
Keeney, R.L., Raiffa, H., 1976. Decision with multiple objectives. Wiley, New York.
Kiker, G.A., Bridges, T.S., Varghese, A., Seager, T.P., Linkov, I., 2005. Application of multicriteria decision analysis in environmental decision making. Integrated Environ. Assess. Manag. 1, 95–108. https://doi.org/10.1897/IEAM_2004a-015.1.
Kottemann, J.E., Davis, F.D., 1991. Decisional conflict and user acceptance of multi-criteria decision-making aids*. Decis. Sci. 22, 918–926. https://doi.org/10.1111/j.1540-5915.1991.tb00371.x.
Kunz, N.C., Moran, C.J., Kastelle, T., 2013. Implementing an integrated approach to water management by matching problem complexity with management responses: a case study of a mine site water committee. J. Clean. Prod. 52, 362–373. https://doi.org/10.1016/j.jclepro.2013.03.018.
Lahdelma, R., Salminen, P., Hokkanen, J., 2000. Using multicriteria methods in environmental planning and management. Environ. Manag. 26, 595–605. https://doi.org/10.1007/s002670010118.
Lai, V.S., Wong, B.K., Cheung, W., 2002. Group decision making in a multiple criteria environment: A case using the AHP in software selection. Eur. J. Oper. Res. 137, 134–144. https://doi.org/10.1016/S0377-2217(01)00084-4.
Langhans, S., Reichert, P., Schuwirth, N., 2014. The method matters: A guide for indicator aggregation in ecological assessments. Ecol. Indicat. 45, 494–507. https://doi.org/10.1016/j.ecolind.2014.05.014.
Langhans, S.D., Lienert, J., 2016. Four common simplifications of multi-criteria decision analysis do not hold for river rehabilitation. PLoS One 11. https://doi.org/10.1371/journal.pone.0150695.
Larsen, T.A., Gujer, W., 1997. The concept of sustainable urban water management. Water Sci. Technol. 35, 3–10.
Lichtenstein, S., Slovic, P., 2006. The construction of preference. Cambridge University Press.
Lienert, J., Larsen, T.A., 2009. High acceptance of urine source separation in seven European countries: a review. Environ. Sci. Technol. 44, 556–566.
Lienert, J., Larsen, T.A., 2007. Soft paths in wastewater management: the pros and cons of urine source separation. GAIA - Ecol. Perspect. Sci. Soc. 16, 280–288.
Lienert, J., Monstadt, J., Truffer, B., 2006. Future scenarios for a sustainable water sector: a case study from Switzerland. Environ. Sci. Technol. 40, 436–442.
Lienert, J., Schnetzer, F., Ingold, K., 2013. Stakeholder analysis combined with social network analysis provides fine-grained insights into water infrastructure planning processes. J. Environ. Manag. 125, 134–148.
Lienert, J., Scholten, L., Egger, C., Maurer, M., 2014. Structured decision-making for sustainable water infrastructure planning and four future scenarios. EURO J. Decis. Process. 1–34.
Linkov, I., Satterstrom, F.K., Kiker, G., Batchelor, C., Bridges, T., Ferguson, E., 2006. From comparative risk assessment to multi-criteria decision analysis and adaptive management: Recent developments and applications. Environ. Int. 32, 1072–1093.
Liu, Y., Gupta, H., Springer, E., Wagener, T., 2008. Linking science with environmental decision making: Experiences from an integrated modeling approach to supporting sustainable water resources management. Environ. Model. Softw. 23, 846–858. https://doi.org/10.1016/j.envsoft.2007.10.007.
Maechler, R.S., Hornik, H., 2016. cluster: Cluster Analysis Basics and Extensions.
Mahmoud, M., Liu, Y., Hartmann, H., Stewart, S., Wagener, T., Semmens, D., Stewart, R., Gupta, H., Dominguez, D., Dominguez, F., others, 2009. A formal framework for scenario development in support of environmental decision-making. Environ. Model. Softw. 24, 798–808.
Mardle, S., Pascoe, S., Herrero, I., 2004. Management Objective Importance in Fisheries: An Evaluation Using the Analytic Hierarchy Process (AHP). Environ. Manag. 33, 1–11. https://doi.org/10.1007/s00267-003-3070-y.
Marttunen, M., Belton, V., Lienert, J., 2018. Are objectives hierarchy related biases observed in practice? A meta-analysis of environmental and energy applications of multi-criteria decision analysis. Eur. J. Oper. Res. 265, 178–194.
Marttunen, M., Hämäläinen, R.P., 2008. The decision analysis interview approach in the collaborative management of a large regulated water course. Environ. Manag. 42, 1026. https://doi.org/10.1007/s00267-008-9200-9.
Marttunen, M., Lienert, J., Belton, V., 2017. Structuring problems for Multi-Criteria Decision Analysis in practice: A literature review of method combinations. Eur. J. Oper. Res. 263, 1–17.
Marttunen, M., Mustajoki, J., Dufva, M., Karjalainen, T.P., 2015. How to design and realize participation of stakeholders in MCDA processes? A framework for selecting an appropriate approach. EURO J. Decis. Process. 3, 187–214.
Massey, A.P., Wallace, W.A., 1996. Understanding and facilitating group problem structuring and formulation: Mental representations, interaction, and representation aids. Decis. Support Syst., DSS Model. Formul. 17, 253–274. https://doi.org/10.1016/0167-9236(96)00004-8.
Matsatsinis, N.F., Samaras, A.P., 2001. MCDA and preference disaggregation in group decision support systems. Eur. J. Oper. Res. 130, 414–429. https://doi.org/10.1016/S0377-2217(00)00038-2.
Mays, N., Pope, C., Popay, J., 2005. Systematically reviewing qualitative and quantitative evidence to inform management and policy-making in the health field. J. Health Serv. Res. Pol. 10, 6–20. https://doi.org/10.1258/1355819054308576.
McCune, B., Grace, J., 2002. Analysis of Ecological Communities. Mjm Software Design, United States.
McDaniels, T.L., Gregory, R.S., Fields, D., 1999. Democratizing risk management: successful public involvement in local water management decisions. Risk Anal. 19, 497–510. https://doi.org/10.1111/j.1539-6924.1999.tb00424.x.
Mendoza, G.A., Martins, H., 2006. Multi-criteria decision analysis in natural resource management: a critical review of methods and new modelling paradigms. For. Ecol. Manag. 230, 1–22.
Mitchell, R.K., Agle, B.R., Wood, D.J., 1997. Toward a theory of stakeholder identification and salience: Defining the principle of who and what really counts. Acad. Manag. Rev. 22, 853–886.
Munda, G., Nijkamp, P., Rietveld, P., 1994. Qualitative multicriteria evaluation for environmental management. Ecol. Econ. 10 (2), 97–112.
Mustajoki, J., Hämäläinen, R.P., Salo, A., 2005. Decision support by interval SMART/SWING—incorporating imprecision in the SMART and SWING methods. Decis. Sci. 36, 317–339. https://doi.org/10.1111/j.1540-5414.2005.00075.x.
Mutikanga, H.E., Sharma, S.K., Vairavamoorthy, K., 2011. Multi-criteria decision analysis: a strategic planning tool for water loss management. Water Resour. Manag. 25, 3947–3969.
Nederhof, A.J., 1985. Methods of coping with social desirability bias: A review. Eur. J. Soc. Psychol. 15, 263–280.
Neuwirth, E., 2014. RColorBrewer: ColorBrewer Palettes.
Novick, E., Senn, D., 2014. External Nutrient Loads to San Francisco Bay (No. Contribution No. 704). San Francisco Estuary Institute, Richmond, CA.
Oksanen, J., Blanchett, G., Michael, Friendly, Roeland, Kindt, Legendre, Pierre, McGlinn, Dan, Minchin, Peter, O'Hara, R.B., Simpson, Gavin, Solymos, Peter, Henry Stevens, M., Szoecs, Eduard, Wagner, Helene, 2017. Vegan: Community Ecology Package.
Payne, J.W., Bettman, J.R., Johnson, E.J., 1992. Behavioral decision research: A constructive processing perspective. Annu. Rev. Psychol. 43, 87–131.
Pearson, L.J., Coggan, A., Proctor, W., Smith, T.F., 2010. A sustainable decision support framework for urban water management. Water Resour. Manag. 24, 363–376.
Pereira, A.G., Munda, G., Paruccini, M., 1994. Generating alternatives for siting retail and service facilities using genetic algorithms and multiple criteria decision techniques. J. Retailing Consum. Serv. 1, 40–47.
Pérez-González, A., Urtiaga, A.M., Ibáñez, R., Ortiz, I., 2012. State of the art and review on the treatment technologies of water reverse osmosis concentrates. Water Res. 46, 267–283.
Peterson, G.D., Cumming, G.S., Carpenter, S.R., 2003. Scenario planning: a tool for conservation in an uncertain world. Conserv. Biol. 17, 358–366. https://doi.org/10.1046/j.1523-1739.2003.01491.x.
Pope, C., Mays, N., 1995. Qualitative research: reaching the parts other methods cannot reach: an introduction to qualitative methods in health and health services research. BMJ 311 (6996), 42–45.
Pöyhönen, M., Hämäläinen, R.P., 2000. There is hope in attribute weighting. INFOR Inf. Syst. Oper. Res. 38, 272–282.
R Core Team, 2013. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
Radjenović, J., Petrović, M., Ventura, F., Barceló, D., 2008. Rejection of pharmaceuticals in nanofiltration and reverse osmosis membrane drinking water treatment. Water Res. 42, 3601–3610. https://doi.org/10.1016/j.watres.2008.05.020.
Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biol. Conserv. 141, 2417–2431.
Reed, M.S., Graves, A., Dandy, N., Posthumus, H., Hubacek, K., Morris, J., Prell, C., Quinn, C.H., Stringer, L.C., 2009. Who's in and why? A typology of stakeholder analysis methods for natural resource management. J. Environ. Manag. 90, 1933–1949. https://doi.org/10.1016/j.jenvman.2009.01.001.
Reichert, P., Langhans, S.D., Lienert, J., Schuwirth, N., 2015. The conceptual foundation of environmental decision support. J. Environ. Manag. 154, 316–332.
Reichert, P., Schuwirth, N., Langhans, S., 2013. Constructing, evaluating and visualizing value and utility functions for decision support. Environ. Model. Softw. 46, 283–291.
San Francisco Bay Nutrient Management Strategy, 2016. Charter of the San Francisco Bay Nutrient Management Strategy: Purpose, Organization, and Governance of the Nutrient Management Strategy.
San Francisco Bay Regional Water Quality Control Board, 2014. Waste Discharge Requirements for Nutrients from Municipal Wastewater Discharges to San Francisco Bay (No. Order No. R2-2014-0014). San Francisco Bay Regional Water Quality Control Board.
San Francisco Public Utilities Commission, 2016. 2015 Urban Water Management Plan for the City and County of San Francisco.
Schoemaker, P.J.H., 1995. Scenario planning: a tool for strategic thinking. Sloan Manag. Rev. Camb. Mass 36, 25–40.
Scholten, L., Maurer, M., Lienert, J., 2017. Comparing multi-criteria decision analysis and integrated assessment to support long-term water supply planning. PLoS One 12. https://doi.org/10.1371/journal.pone.0176663.
Scholten, L., Scheidegger, A., Reichert, P., Maurer, M., 2013. Combining expert knowledge and local data for improved service life modeling of water supply networks. Environ. Model. Softw. 42, 1–16. https://doi.org/10.1016/j.envsoft.2012.11.013.
Scholten, L., Schuwirth, N., Reichert, P., Lienert, J., 2015. Tackling uncertainty in multi-criteria decision analysis–An application to water supply infrastructure planning. Eur. J. Oper. Res. 242, 243–260. https://doi.org/10.1016/j.ejor.2014.09.044.
Schuwirth, N., Reichert, P., Lienert, J., 2012. Methodological aspects of multi-criteria decision analysis for policy support: A case study on pharmaceutical removal from hospital wastewater. Eur. J. Oper. Res. 220, 472–483. https://doi.org/10.1016/j.ejor.2012.01.055.
Scott, C.A., Bailey, C.J., Marra, R.P., Woods, G.J., Ormerod, K.J., Lansey, K., 2012. Scenario planning to address critical uncertainties for robust and resilient water–-wastewater infrastructures under conditions of water scarcity and rapid development. Water 4, 848–868.
Simpson, G., 2016. Permute: Functions for Generating Restricted Permutations of Data.
Soltani, A., Hewage, K., Reza, B., Sadiq, R., 2015. Multiple stakeholders in multi-criteria

decision-making in the context of municipal solid waste management: a review. Waste Manag. 35, 318–328.
Starkl, M., Brunner, N., Flögl, W., Wimmer, J., 2009. Design of an institutional decision-making process: The case of urban water management. J. Environ. Manag. 90, 1030–1042. https://doi.org/10.1016/j.jenvman.2008.03.012.
Stokes, J.R., Horvath, A., 2009. Energy and air emission effects of water supply. Environ. Sci. Technol. 43, 2680–2687. https://doi.org/10.1021/es801802h.
Störmer, E., Truffer, B., 2009. Strategic decision making in infrastructure sectors: participatory foresight and strategic planning for sustainable sanitation. Geograph. Helv. 64, 73–80.
Störmer, E., Truffer, B., Dominguez, D., Gujer, W., Herlyn, A., Hiessl, H., Kastenholz, H., Klinke, A., Markard, J., Maurer, M., others, 2009. The exploratory analysis of trade-offs in strategic planning: Lessons from Regional Infrastructure Foresight. Technol. Forecast. Soc. Change 76, 1150–1162.
Sutula, M., Senn, D., 2015. Scientific Basis to Assess the Effects of Nutrients on San Francisco Bay Beneficial Uses (Technical Report 864 No. Contract 11-151-120). Prepared for the San Francisco Bay Regional Water Quality Control Board.
Tarpeh, W.A., Udert, K.M., Nelson, K.L., 2017. Comparing ion exchange adsorbents for nitrogen recovery from source-separated urine. Environ. Sci. Technol. 51, 2373–2381. https://doi.org/10.1021/acs.est.6b05816.
Thornton, J., Harding, W., Dent, M., Hart, R., Lin, H., Rast, C., Rast, W., Ryding, S., Slawski, T., 2013. Eutrophication as a 'wicked' problem. Lakes Reservoirs Res. Manag. 18, 298–316. https://doi.org/10.1111/lre.12044.
Trautmann, H., Steuer, D., Mersmann, O., Bornkamp, B., 2014. truncnorm: Truncated normal distribution.
Truffer, B., Störmer, E., Maurer, M., Ruef, A., 2010. Local strategic planning processes and sustainability transitions in infrastructure sectors. Environ. Policy Gov. 20, 258–269.
Umar, M., Roddick, F., Fan, L., 2015. Recent advancements in the treatment of municipal wastewater reverse osmosis concentrate—an overview. Crit. Rev. Environ. Sci. Technol. 45, 193–248.
Van Dolah, F.M., 2000. Marine algal toxins: origins, health effects, and their increased occurrence. Environ. Health Perspect. 108, 133.
Vazquez-Montiel, O., Horan, N.J., Mara, D.D., 1996. Management of domestic wastewater for reuse in irrigation. Water Sci. Technol., Wastewater Reclam. Reuse 33, 355–362. 1995. https://doi.org/10.1016/0273-1223(96)00438-6.
Vos, J.F.J., Achterkamp, M.C., 2006. Stakeholder identification in innovation projects: Going beyond classification. Eur. J. Innov. Manag. 9, 161–178. https://doi.org/10.1108/14601060610663550.
Wankel, S.D., Carol, K., Chris, F.A., Paytan, A., 2005. Nitrogen sources and cycling in the San Francisco Bay Estuary: A nitrate dual isotopic composition approach. AGU Fall Meet. Abstr. 1, 0982.
Wickham, H., 2011. The split-apply-combine strategy for data analysis. J. Stat. Softw. 40, 1–29.
Wilkinson, A., Kupers, R., 2014. The Essence of Scenarios: Learning from the Shell Experience. Amsterdam University Press, Amsterdam.
Wren, I., 2017. Treatment Wetlands for Nutrient Removal from Bay Area Wastewater Facilities: Screening Level Opportunities and Constraints Analysis - Draft Report. San Francisco Bay Nutrient Management Strategy, Oakland, CA.
Wright, G., Goodwin, P., 2009. Decision making and planning under low levels of predictability: Enhancing the scenario method. Int. J. Forecast., Specl. Sect. Decis. Mak. Plann. Under Low Levels Predict. 25, 813–825. https://doi.org/10.1016/j.ijforecast.2009.05.019.
Zahir, S., 1999. Clusters in a group: Decision making in the vector space formulation of the analytic hierarchy process. Eur. J. Oper. Res. 112, 620–634.
Zheng, J., Egger, C., Lienert, J., 2016. A scenario-based MCDA framework for wastewater infrastructure planning under uncertainty. J. Environ. Manag. 183, 895–908.
Zuur, A., Ieno, E., Smith, G., 2007. Analyzing Ecological Data. Springer Science + Business Media, New York.