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Meeting the Growing Need for Land-Water System Modelling to Assess Land Management Actions

Scott T. Larned[1] · Ton H. Snelder[2]

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Abstract

Elevated contaminant levels and hydrological alterations resulting from land use are degrading aquatic ecosystems on a global scale. A range of land management actions may be used to reduce or prevent this degradation. To select among alternative management actions, decision makers require predictions of their effectiveness, their economic impacts, estimated uncertainty in the predictions, and estimated time lags between management actions and environmental responses. There are multiple methods for generating these predictions, but the most rigorous and transparent methods involve quantitative modelling. The challenge for modellers is two-fold. First, they must employ models that represent complex land-water systems, including the causal chains linking land use to contaminant loss and water use, catchment processes that alter contaminant loads and flow regimes, and ecological responses in aquatic environments. Second, they must ensure that these models meet the needs of endusers in terms of reliability, usefulness, feasibility and transparency. Integrated modelling using coupled models to represent the land-water system can meet both challenges and has advantages over alternative approaches. The need for integrated land-water system modelling is growing as the extent and intensity of human land use increases, and regulatory agencies seek more effective land management actions to counter the adverse effects. Here we present recommendations for modelling teams, to help them improve current practices and meet the growing need for land-water system models. The recommendations address several aspects of integrated modelling: (1) assembling modelling teams; (2) problem framing and conceptual modelling; (3) developing spatial frameworks; (4) integrating economic and biophysical models; (5) selecting and coupling models.

Keywords Aquatic ecosystems · Integrated environmental modelling · Land-water systems · Land management actions · Regulatory processes · Scenarios

Introduction

The global extent of land conversion (e.g. from natural forest and grassland to agricultural and urban land) and rates of land use intensification (e.g. fertiliser input, crop yields, urban infilling) have increased roughly monotonically for at least a century, and human land use now accounts for over 70% of the Earth's ice-free land surface (Radwan et al. 2021; Kastner et al. 2022; Potapov et al. 2022). Under some socioeconomic and climate scenarios, land conversion and land-use intensification are projected to continue into the late 21st Century (Chen et al. 2020; Pal et al. 2021). Based on this massive, ongoing transformation, land use is inarguably a major driver of global changes ranging from growth in food production, human populations and prosperity, to water scarcity, water, air and soil pollution, greenhouse gas emissions and biodiversity loss (Bai et al. 2017; Ramankutty et al. 2018).

Among the many environmental effects of human land use are increases in the loads of nutrients, biocides, sediment and other contaminants in drainage networks, and altered hydrological regimes (Seitzinger et al. 2010; Ippolito et al. 2015; Bosmans et al. 2017; Borrelli et al. 2020). In turn, elevated contaminant loading and hydrological alterations contribute to the degradation of aquatic ecosystems, including rivers, lakes, estuaries and coastal zones (Meybeck 2004; Freeman et al. 2019; Wurtsbaugh et al. 2019). Managing land to reverse this degradation is both a societal

✉ Scott T. Larned
scott.larned@niwa.co.nz

1 National Institute of Water and Atmospheric Research, Christchurch, New Zealand

2 LWP Ltd, Christchurch, New Zealand

imperative and a 'wicked problem' in the sense that conflicts between environmental and socioeconomic values, limited knowledge of land use effects, and high risks of unintended consequences preclude simple solutions (Patterson et al. 2013; DeFries and Nagendra 2017).

Management actions intended to reduce and reverse the adverse effects of land use on aquatic ecosystems fall into two broad classes: land use regulation (e.g. land use zoning, nutrient input/output controls, water use limits) and the adoption of mitigation systems (e.g. erosion control, riparian planting, stormwater infiltration systems). We refer to these approaches collectively as 'land management actions', and to combinations of prospective land management actions as 'land management scenarios'. Ideally, land management actions benefit aquatic ecosystems while minimising compliance costs and constraints on land use (Doole and Romera 2014; Rolfe and Gregg 2015). Stakeholders and decision-makers in land and water management generally have multiple scenarios to consider, ranging from 'business as usual' (no new action) to ambitious scenarios involving numerous and/or extensive management actions (Quinn et al. 2013; Reed et al. 2013; Zhang et al. 2017).

Decisions regarding land management actions are based in large part on assessments of alternative actions and in particular their predicted environmental benefits and socioeconomic impacts. Decision-makers and stakeholders rely on environmental and economic science for these assessments. The demand for science-informed land management actions is increasing rapidly, driven by the recognition that contaminant loading and hydrological alteration associated with land use is exceeding the assimilative capacities of aquatic receiving environments (Diamond et al. 2015; Vilmin et al. 2018). Despite the reliance on scientific predictions, there is also scepticism about those predictions and the tools used to produce them (Addison et al. 2013; DeAngelis et al. 2021). This scepticism reflects concern about the reliability (or conversely, the uncertainty), utility and relevance of scientific predictions, the time and expense required to produce them, and the perception that predictive tools are overly complicated and lack transparency.

Quantitative modelling can provide decision-makers with information that is timely, useful for the decision-making process, and relevant to specific land and water management problems. Less rigorous methods such as unaided expert opinions and cross-site comparisons are frequently used in lieu of modelling to save time and reduce costs, but these methods rarely account for the inherent complexity of biophysical and economic systems or provide mechanistic explanations for responses to management actions (Sutherland 2006; Cook et al. 2010; Addison et al. 2013). Expert opinion is particularly popular for assessing alternative land management actions because it is fast and requires minimal data (Bohnet et al. 2011; Dunn et al. 2015). However, forming opinions is not a transparent process and expert opinions are not amendable to uncertainty or sensitivity analyses, or evaluations of their underlying assumptions. In contrast, the structured process of developing quantitative models can, if done properly, make them both transparent and highly amendable to analyses and evaluation (Fisher et al. 2010; Grimm et al. 2014; Jones et al. 2020). Quantitative models also involve subjective decisions that affect model output (Krueger et al. 2012; Hämäläinen and Lahtinen 2016), but those decisions can be made transparent and testable.

In this paper, we first make a case for integrated environmental modelling as a rigorous and transparent approach for assessing land management actions. Next, we discuss the role of modellers and land-water system models in highly scrutinised regulatory processes. We then provide recommendations for environmental modelling teams, to help them meet the growing need for quantitative modelling to inform land management actions.

Modelling to Support the Assessment and Selection of Land Management Actions

Informed decisions about land-water management problems require an understanding of the effects of current land use and predictions about the effectiveness of prospective land management actions. Meeting these requirements is not a simple matter of correlating land-based activities with conditions in aquatic ecosystems, because the former are rarely the proximate determinants of the latter. Instead, the effects of land-based activities on aquatic environments are predominately indirect and mediated through 'causal chains' of interlinked biophysical processes, including contaminant generation and loss from land, transport and transformation in drainage networks, and input to and ecological responses within aquatic receiving environments (Burcher et al. 2007; Allan et al. 2012). In this paper, we focus on chains of hydrologically linked processes that are distributed across catchments, from sites on land where contaminants originate to the aquatic receiving environments where ecological objectives are to be achieved (Larned et al. 2020, 2022) and refer to them collectively as 'land-water systems' (Snelder et al. 2022).

Predictions of the effects of land management actions need to account for the spatial and biophysical complexity of land-water systems and may need to be projected into the future, to characterise trajectories of environmental responses to prospective actions (Law et al. 2017). In particular, endusers need estimates of lag times, i.e., response delays due to slow processes such as contaminant transport in groundwater and internal loading in lakes (Meals et al. 2010; Melland et al. 2018; Abell et al. 2022). Lag time estimates can forestall unrealistic expectations of rapid responses, thereby maintaining long-term support for land management actions (Ascott et al. 2021). Projections about land management actions may also need to account for future changes in causal factors other than land use, such as climate variability and climate change (Ockenden et al. 2017; Xu et al. 2018).

The appropriate tools for the tasks set out here are quantitative models that can simulate land-water systems, project responses to management actions and other future changes, and estimate lag times and prediction uncertainty. The process of formulating and assessing land management actions is usually an iterative one: one or more land management scenarios are identified; simulation modelling is carried out and the outputs are interpreted and assessed; the assessments are used to reformulate or modify the scenarios and the process is repeated. This cycle is used to triangulate on and select the most acceptable management actions for implementation (Liu et al. 2008a). The use of quantitative, process-based models in this iterative process allows modellers to explain why their models produce different outcomes for different scenarios, i.e., how different land management actions cause different responses in catchment processes, which in turn cause different responses in receiving environments (Cuddington et al. 2013, Black et al. 2014). These differences generally need to be expressed in terms of continuous variables, which also requires quantitative modelling.

Appropriate models of land-water systems are 'holistically fit-for-purpose' (Hamilton et al. 2022), where fitness is based on a wide range of model attributes such as accuracy, development costs, timeliness, and the relevance of outputs to stakeholder needs. Numerous criteria for evaluating model fitness have been proposed, including relevance, validity, and salience (Van Voorn et al. 2016; Eker et al. 2018; Hamilton et al. 2019, 2022). Three general criteria recommended by Hamilton et al. (2022)—usefulness, reliability and feasibility—encompass most of the previously proposed criteria. Usefulness refers to the degree to which a model can respond to questions posed by endusers. Reliability refers to the scientific validity of a model, its perceived credibility, and its treatment of prediction uncertainty. Feasibility refers to the degree to which modelling can be carried out within the practical constraints of deadlines, limited funding and data shortages. Usability, reliability and feasibility are not mutually exclusive and trade-offs between them are inevitable (Cash et al. 2002; Van Voorn et al. 2016). For example, minimising development time to increase feasibility is likely to curtail model testing, which reduces reliability. To frame our recommendations for developing land-water system models, we refer to usability, reliability and feasibility as defined by Hamilton et al. (2022), plus a fourth criterion, transparency. Transparency refers to the degree to which modellers document their procedures and assumptions, report and explain the implications of model performance, limitations and uncertainties, and ensure that the modelling process can be independently verified (Jakeman et al. 2006; Schmolke et al. 2010; Grimm et al. 2014, Planque et al. 2022). Transparency should be seen as a fundamental criterion for environmental modelling and not subject to trade-offs.

Integrated Environmental Modelling

Environmental modellers have traditionally worked in discipline-specific groups, in which rapid model development is made possible by sharing specialist knowledge and methods. Hydrological, ecological and catchment water quality modellers are typical of these groups, which produce discipline-specific models. However, the discipline-based approach to modelling is inadequate for developing models of land-water systems, predicting the effects of alternative land management scenarios, and conveying the results to endusers in an intelligible and usable format. Modelling land-water systems requires an interdisciplinary approach comprised, at a minimum, of hydrological, catchment water quality and ecological modelling expertise. For many applications, economic modelling is also needed to inform endusers about the socio-economic implications of land management actions, as discussed below and in Section 5. Academic work on models that simulate land use, contaminant loading and aquatic ecosystem responses began several decades ago (e.g. Wilkinson et al. 1997). Progress was initially hindered by the compartmentalisation noted above, and by data shortages and limited computing power. In the last two decades, these impediments have been reduced through the formation of interdisciplinary modelling groups, libraries of open-source models, access to high-performance computers, and rapid growth in data generation (Li et al. 2018; Gao et al. 2019). These changes have led to a proliferation of modelling projects and rapid growth in scientific publications (Zare et al. 2017; Soares and Calijuri 2021).

Early efforts to model land-water systems relied on monolithic "integral" models in which the entire causal chain was represented by a single, complex model (Lu and Piasecki 2012, Voinov and Shugart 2013). Most of the monolithic models were proprietary and designed to simulate specific systems; modifications for other purposes required specialist skills and agreement by the licensor. This lack of flexibility and their costly and slow development has limited the usefulness and feasibility of monolithic models in land and water management and decision-making.

Assemblies of stand-alone environmental models were first trialled in the 1980s as an alternative to monolithic

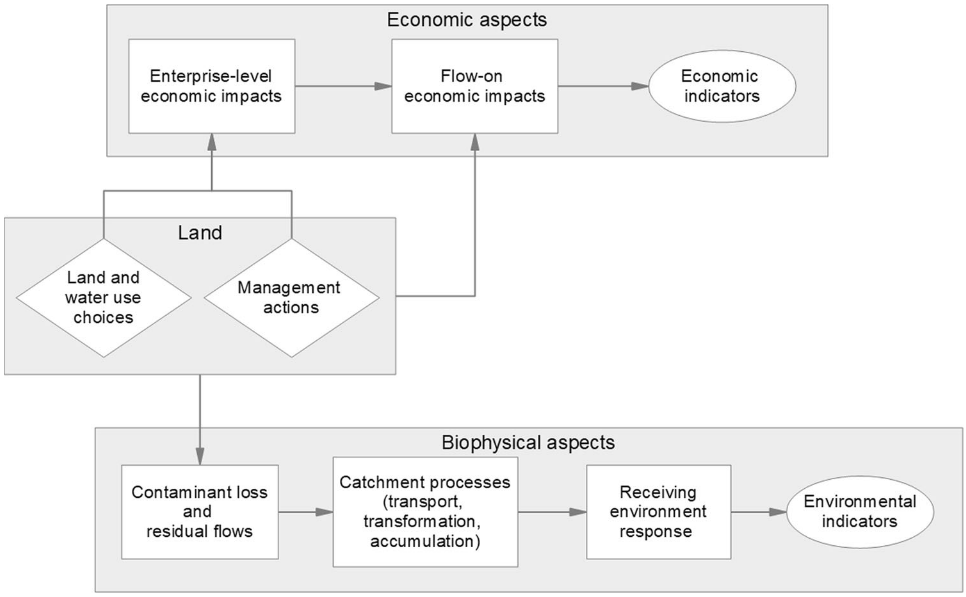

Fig. 1 Schematic diagram of a land water system model. Central box: modelled representations of land in the model domain, incorporating land and water use and management actions. Diamonds: component models that can be manipulated to simulate choices of land use, associated water use, and land management actions. Upper and lower boxes: economic and biophysical aspects of the land water system model. Rectangles: component models that simulate a chain of economic and biophysical processes. Arrows: model outputs being passed to the next component model. Ovals: indicators of economic and environmental impact that are provided to endusers. Indicators can be based on output from any of the component models, not only the final component model

models (Moore and Hughes 2017). The term 'integrated environmental modelling' (IEM) is now widely used to describe this approach. At its most basic, IEM uses chains of two or more component models in which output data from one model provides input data to the next model in the chain. Advantages of IEM over monolithic models include shorter development times, a wealth of shared source code and model frameworks, and most important, the large number of possible model combinations, which provides the flexibility needed to address a wide range of environmental management issues. In many cases, IEM projects use extant models to avoid the time-consuming creation of new, bespoke models. Repurposing and recombining extant models to meet different needs are now hallmarks of IEM (Belete et al. 2017; Elsawah et al. 2017).

IEM has been used to address environmental problems ranging from climate change adaptation to radiation risk assessments to water resources planning (Laniak et al. 2013; Moore and Hughes 2017). Land-water management problems are a subset of the larger problem set, and we refer to integrated models used to address these problems as 'land-water system' (LWS) models.

LWS models typically represent the following chain of biophysical processes: (1) land use and land management practices; (2) contaminant mobilisation and loss from land; (3) contaminant transport through drainage networks to receiving environments (and associated contaminant transformation, removal and transient storage); (4) hydrodynamics and contaminant mixing in the receiving environment; (5) direct ecological responses to contaminant exposure (e.g. phytoplankton production), and in some cases, indirect effects mediated by trophic interactions and other ecosystem processes (e.g. Trolle et al. 2014; Johnston et al. 2017; Crossman, Elliott (2018)). LWS models can also be used to assess the effects of hydrological alterations associated with land use, such as water abstraction, impoundment and irrigation (e.g. Casper et al. 2011; Guse et al. 2015).

For formulating and assessing land management actions, LWS models usually start with alternative land use scenarios for the model domain (Snelder et al. 2022; Fig. 1). The scenarios can represent different land use choices, different regulations (e.g. input controls, zoning), and different mitigation systems (e.g. treatment wetlands, stormwater detention basins). Using alternative scenarios, endusers can ask 'what-if' questions, compare the predicted outcomes of different management actions, and evaluate trade-offs between those outcomes (Liu et al. 2008b; Voinov et al. 2016).

For many LWS model applications, economic modelling is needed to inform endusers about the economic implications of land management actions. In these applications, the LWS model must include economic drivers, processes and responses (e.g. Kragt et al. 2011, Qureshi et al. 2013; Weng et al. 2020). The economic aspects of land-water systems are typically represented in an LWS model by the economic benefits of land use choices and the costs associated with management actions (Fig. 1). The model needs to simulate these benefits and costs for enterprises (e.g. farm businesses) and aggregate them across modelled catchments. Simulations may include the flow-on impacts of land management actions on businesses and institutions that are not directly involved in land use (e.g. service industries). The economic impacts of land management actions need to be predicted for two reasons. First, the costs of land management actions generally increase with increasing levels of environmental protection, and endusers need information about both costs and protection to understand the trade-offs and make normative decisions (Roebeling et al. 2015; Plunge et al. 2023). Second, multiple land management actions with different costs can potentially achieve the same environmental objective, and land management decisions often comprise mixtures of actions (Goeller et al. 2020). From a decision maker's perspective, the optimal mix of actions is one that best balances the overall costs and overall protection.

To evaluate trade-offs between the environmental benefits and economic impacts of management actions, decision-makers may need the output from LWS models to be distilled into 'indicators' (Hamilton et al. 2015). Environmental and economic indicators (e.g. biodiversity, trophic state, employment, household income) are calculated from model output, and used by decision-makers to compare land management scenarios. The selection of indicators is a shared task for endusers and modellers; the former group needs to identify contextually relevant indicators, and the latter group needs to develop models that can quantify those indicators.

Land-Water System Models in Environmental Regulatory Processes

Arguably, the greatest needs for LWS models are associated with environmental regulatory processes, i.e., planning and decision-making in the context of environmental policy development and implementation (Holmes et al. 2009; Fisher et al. 2010; DeAngelis et al. 2021). Regulatory processes are often contentious, due to conflicts between stakeholders' environmental and socio-economic values, and the need to allocate limited environmental resources, such as the assimilative capacity of aquatic environments (Rees 2017). Environmental models used in regulatory processes are subject to high levels of scrutiny. To withstand this scrutiny, the models must be highly transparent, and the inevitable trade-offs between model usefulness, reliability and feasibility must be justified (Fisher et al. 2010; Özkundakci et al. 2018).

In this section, we discuss the use of LWS models and model output by the different groups involved in environmental regulatory processes. We use a generalised organisational framework to show the relationships between these groups but note that the lines of communication and their content and detail are the important points, not the organisational structure per se. Proponents of LWS modelling often recommend an inclusive approach termed 'participatory modelling', which can include participants from regulatory agencies, interest groups, academics, local government and the general public (Paolisso et al. 2015; Glynn 2017; Voinov et al. 2016, 2018). In our view, input from a wide range of participants can be valuable during the initial problem scoping and conceptual modelling stages of LWS projects, as discussed in Section 5B below. However, the inclusive approach rarely applies to the subsequent stages of quantitative model development and scenario simulation. Participation in these stages of the regulatory process is generally limited to modellers but the outputs are relevant to all participants.

Regulatory processes aim to address land-water management problems by mandating land management actions to restrict the effects of land use to acceptable levels while minimising socio-economic impacts. These processes involve intensive discussion and negotiations between regulators and stakeholders aimed at identifying acceptable land management actions. In many cases there are disagreements between different stakeholder groups about the proposed solutions and decision-makers must then act as adjudicators, often using judicial systems to complete the process.

A generalised organisational framework in which environmental regulatory processes take place is shown in Fig. 2. There are three groups in this framework: regulatory agencies, stakeholders and decision-makers. Regulatory agencies are mandated to drive regulatory processes and are responsible for proposing land management actions, and for producing evidence that these will resolve land-water management problems in a fair and equitable way. Regulatory agencies use technical experts (i.e., modelling teams) to provide the substantive information that informs the management actions and policy developers, including planners and lawyers, to formulate the policies and regulations that give the actions their required legal status. These roles may be filled by regulatory agency staff or by consulting modellers, lawyers, and other experts on behalf of the agency. An LWS modelling team typically comprises a lead modeller and specialist modellers (e.g. hydrological, ecological and economic modellers). The lead modeller has overall responsibility for designing and assembling the LWS model and is the primary interface between the modelling team and other groups.

The formation of modelling teams is discussed in detail in Section 5A below. Stakeholder groups also employ technical representatives, along with policy analysts and lawyers to represent their interests (Fig. 2). Stakeholder technical representatives have parallel skills to regulatory

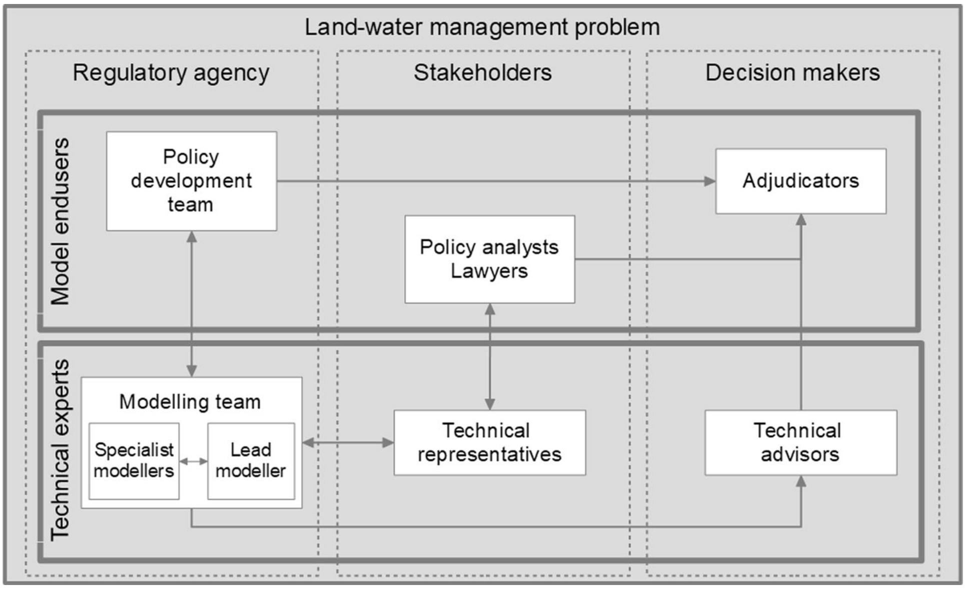

Fig. 2 Schematic diagram describing the organisation of different groups involved in regulatory processes that use environmental models to address land-water management problems. Dashed rectangles differentiate the three types of organisations involved in the regulatory process: regulators, stakeholders and decision-makers. Solid rectangles differentiate technical experts from model endusers. White boxes represent groups with specific roles in regulatory processes. Arrows between boxes represent the principal communication channels between groups. Double-headed arrows represent two-way information exchange. Single-headed arrows indicate that most information is conveyed in one direction

agency modelling teams, and stakeholder policy analysts and lawyers have parallel skills to regulatory agency policy development teams. The decision-making group is represented by technical advisors and adjudicators; the former provides decision-makers with impartial opinions on technical aspects of the proposed solutions, and the latter is mandated to make normative judgements concerning the most appropriate solutions.

We distinguish between technical experts and model endusers in the organisational framework (Fig. 2). Technical experts have a detailed understanding of the modelling process and component models, and can make informed judgements concerning model usefulness, reliability and feasibility. In contrast, model endusers have expertise in environmental management including development of prospective land management actions and formulation of policies and regulations. Model endusers do not need to have modelling expertise, but do need to understand the capabilities and limitations of models, and work with model output.

Key lines of communication are indicated in Fig. 2. Regulatory agency modelling teams work with the policy development teams from the same agencies to characterise land-water management problems and generate potential solutions through iterative scenario analyses. The modelling team also interacts closely with stakeholders, primarily through their technical representatives, who in turn communicate with the broader group of stakeholders they represent. Most stakeholders lack the expertise needed to judge model usefulness and reliability and must rely on their technical experts. Finally, information exchanged between modelling teams and decision-makers is mediated by policy analysts, technical advisors and other intermediaries.

It is clear from the preceding discussion that the groups involved in regulatory processes have distinct and disparate knowledge and skills, which can hinder communication. Knowledge brokering is an essential function for facilitating communication between these groups. The use of knowledge brokers in LWS modelling projects (and applied environmental modelling in general) is widely promoted (e.g. Kragt et al. 2013, Cvitanovic et al. 2016). Many of these references portray knowledge brokers as communication specialists who are separate from the technical experts and model endusers. However, communication specialists with sufficient modelling and environmental management expertise to be effective knowledge brokers are rare. In our view, knowledge brokering is a shared responsibility across all groups in Fig. 2.

Recommendations for Increasing the Effectiveness and Efficiency of Land-Water System Modelling

In this section, we set out recommendations aimed at improving the effectiveness of LWS modellers in their roles in environmental regulatory processes, and at increasing modelling efficiency in terms of time and costs. We note that many other sources of guidance for environmental modelling are available, including technical guidance and non-technical, 'behavioural' guidance for working with stakeholders (e.g. Alexandrov et al. 2011; Moriasi et al. 2015; Voinov et al. 2016). Most of the previously published guidance is generic and pertains to all applied environmental modelling projects. Here we are concerned with issues that are specific to integrated LWS modelling, for which practical guidance is rare.

For several recommendations, we use the Chesapeake Bay Programme modelling system as an exemplar (Hood et al. 2021, Kaufman et al. 2021). This programme is widely regarded as one of the most successful model-guided environmental management initiatives in the world (Boesch 2019). In general terms, the Chesapeake Bay Program uses a suite of coupled biophysical models for simulating nutrient and sediment losses and loads in the Chesapeake Bay catchment and ecosystem responses in the Bay, plus scenario assessment tools for evaluating the ecological and economic effects of prospective changes in land and air quality management. The modelling and scenario analysis system has been used to inform the development of environmental regulations in the catchment for several decades. Reductions in contaminant loads and improvements in aquatic ecosystem health following the enactment of these regulations suggests that they are having the intended effects, which lends support the use of LWS models for regulatory processes (Lefcheck et al. 2018).

Assembling and Leading Modelling Teams

Our first recommendations concern assembling and leading modelling teams with the collective skills needed to produce integrated LWS models and to work constructively with endusers. For modelling land-water systems, expertise is required in modelling land use, contaminant loss, catchment hydrology and water quality, hydraulics, ecological processes, and in many cases, economics. In addition to technical expertise, important qualities in team members are a willingness to collaborate, a problem-solving mindset and a commitment to impartiality. The interdisciplinary nature of LWS modelling and the need to interact with endusers in the regulatory process makes collaboration a necessity. Skills and aptitudes associated with successful collaborations include relationship building, open-mindedness and self-awareness about personal biases and motivations (Glynn 2017; Merritt et al. 2017). A problem-solving mindset in the environmental modelling context refers to the ability to progress under constraints imposed by trade-offs between usefulness, reliability and feasibility. Problem solving involves the pragmatic application of knowledge rather than the discovery of new knowledge, where scientists are less compelled to make such trade-offs (Hulme 2014; Cooke et al. 2020). While regulatory processes are typically contentious and involve stakeholders with vested interests in the outcomes, the modellers must strive for neutrality. Neutrality in the modelling team contributes to the perceived reliability of their models, and fosters trust in the model output (Hamilton et al. 2022).

The role that LWS modellers play in the regulatory process is that of 'honest brokers' in policy development (Pielke 2007). The honest broker role entails conveying information about the likely outcomes of different policy options (e.g. different land management scenarios), and the associated assumptions and uncertainty. Effective honest brokers are characterised by their willingness to engage extensively with stakeholders and decision-makers, their provision of objective information, and their candour and open-mindedness (Gluckman et al. 2021). These characteristics are consistent with the qualities we recommended for LWS modellers.

We recommend that LWS modelling projects have a lead modeller to take overall responsibility for model design and scenario simulations, and for communicating the outputs to endusers. The lead modeller must understand the knowledge, ontologies and terminologies of the different science disciplines involved in the project and have expertise in quantitative modelling and model integration. In addition, a lead modeller needs a good understanding of the regulatory process, and good communication skills to broker knowledge between the modelling team and the policy development team, stakeholders and decision-makers (Fig. 2). There is an equally important leadership function associated with the policy development team. To ensure that the regulatory process is carried out efficiently and effectively, the activities of the modelling team need to be coordinated with those of the policy development team; this coordination is a joint responsibility for the modelling and policy development team leaders.

We note that, while leadership is important for the success of modelling in regulatory processes, we are not recommending that lead modellers have an unreasonable level of authority. Regulatory processes comprise too many different functions for an authoritarian approach; instead, the leadership approach needs to emphasise facilitation and relationship building and act altruistically (i.e., provide guidance to modellers and endusers with the goal of meeting the needs of decision-makers).

Problem Framing and Conceptual Modelling

Problem framing and conceptual modelling are early stages of LWS model development and are characterised by intensive interaction between endusers and modellers. The most detailed interactions are likely to be between

modelling teams and the technical experts employed by stakeholders and decision-makers, but it is essential to convey information to and from all enduser groups.
The aim of problem framing is to reach a shared understanding of the land-water management problem, the stakeholders' objectives, and how the problem and objectives can be structured and delimited in a way that makes them tractable for modelling (Hamilton et al. 2015; Voinov et al. 2016). The problem framing stage is also used to elicit information from stakeholders (e.g. historical observations, community values, acceptable and unacceptable land management actions).

Problem framing is challenging due to the volume and diversity of information conveyed, language barriers (including abstract concepts such as uncertainty and causality), and differences in culture and knowledge between endusers and modellers (Cvitanovic et al. 2016). There is a natural tension between the need to limit the scope of a modelling problem (in order to reduce data, time and funding requirements), and the need to expand the scope (to account for a wide range of stakeholder concerns). In addition to these generic challenges, problem framing in LWS modelling projects has an additional degree of difficulty because it must consider two distinct and inherently complex systems—biophysical systems comprising catchments and aquatic receiving environments, and socioeconomic systems (Geary et al. 2020).

Despite the challenges, collaborative problem framing is vital for establishing trust in the modelling process, developing a work programme, and maximising the chances that model output will be deemed fit-for-purpose and used. Surveys of modellers and endusers indicate that collaborative problem framing is a primary determinant of subsequent model uptake (Merritt et al. 2017; Will et al. 2021). The Chesapeake Bay Program exemplifies the roles of stakeholders in problem framing and work programme design, both of which have evolved over the life of the 40-year-old programme. Multiple stakeholder workgroups with expertise in water quality, agriculture, urban stormwater, forestry, modelling and other areas were established to help modelling teams determine the overall structure and applications of the modelling system, including land management scenario development (Kaufmann et al. 2021). These workgroups are composed of representatives from government agencies with jurisdiction in the catchment, government research agencies, non-governmental organisations, commercial sectors and academia. Problem framing is based on knowledge elicitation, for which there is a wealth of guidance (Voinov et al. 2018). However, elicitation alone is insufficient; the knowledge exchanged among groups must be mutually intelligible. For many stakeholders and decision-makers, technical information about biophysical processes and model development is initially unintelligible. Similarly, legal information and terminology used by policy analysts and lawyers may not be understood by modellers. If left unresolved, these barriers to knowledge exchange can lead to diverging views of the land-water management problem and its potential solutions (Argent et al. 2016). Skilled translation is needed to ensure that the entire problem-framing group eventually reaches a shared perspective; unfortunately, translation between the scientific and regulatory domains is not a widely recognised or valued competency (Schwartz et al. 2017).

We have three recommendations for LWS modelling teams to ensure that the aims of problem framing are achieved. First, plan the problem framing work thoroughly before it commences. Problem framing is often carried out in loosely structured workshops with open ended agendas. In some cases, these workshops are run by generalist facilitators who are unfamiliar with environmental modelling and regulatory processes and cannot mediate conflicting objectives and perspectives. The combination of inadequate planning and weak leadership can result in polarised views about technical issues, which can in turn lead to disengagement by endusers and lost credibility for the modelling process (Voinov et al. 2018, Hedelin et al. 2021). We recommend running more structured problem framing workshops that include an explicit set of outcomes to be reached, clearly articulated methods to elicit knowledge, tools for organising and mapping information (e.g. causal diagrams) and, if necessary, mediation as a means of forging agreements among stakeholders (Voinov and Gaddis 2008; Metcalf et al. 2010; Hamilton et al. 2015).

Second, instead of generalist facilitators, we recommend that workshops are led collaboratively by the individuals who have overall leadership of the regulatory process. Collectively, this group should have a thorough understanding of problem scoping, environmental modelling, land-water management problems and regulatory processes, and the leadership skills needed to help participants stay on track and manage conflicts (Kragt et al. 2013). Here, we differ from authors who consider modellers to be poor prospects for leading collaborative processes, due to their presumed narrow focus on technical work and lack of leadership training (e.g. Hämäläinen et al. 2020). In reality, many environmental modellers have extensive experience working with stakeholders, policy developers and decision-makers, and technical expertise does not preclude skilled leadership.

Third, we recommend that modelling teams employ the knowledge brokering function described in Section 4. The aim of knowledge brokering in the problem scoping stage is to ensure that the modelling team and model enduser groups receive the information that they need about the land-water management problem and prospects for LWS modelling, at the right level of detail. Technical information needs to be translated to suit the level of knowledge of each group and its role in regulatory processes. There is an implicit assumption in some articles on participatory modelling that all enduser groups will have a thorough understanding of environmental modelling and a sustained engagement in model development (Fisher et al. 2010). This assumption is incorrect; with the exception of their technical experts, endusers are likely to have limited knowledge of modelling methods and limited time to engage. More importantly, different endusers have different roles for which they require information to be contextualised. For example, policy writers and policy analysts may need prospective land management actions (e.g. wetland construction, erosion control) expressed in terms of regulated activities (e.g. earthworks, water diversion and impoundment).

Conceptual models in IEM are qualitative descriptions of the environmental systems that are to be simulated with quantitative models. In the context of land-water systems, conceptual models need to include the spatial entities in the model domain (e.g. catchment and sub-catchment boundaries, drainage network, aquatic receiving environments), driver or causal variables (i.e., land uses), catchment processes, response variables in receiving environments, and the prospective management actions and scenarios to be evaluated. Graphical tools and analytical techniques for developing conceptual models are widely available (Argent et al. 2016; Voinov et al. 2018). The initial conceptual model can be altered or supplemented with additional detail as the project progresses.

Conceptual model development with endusers is often omitted to save time and funds for quantitative modelling work, or because the appropriate conceptual model structure appears to be obvious. However, we strongly recommend retaining this step. It serves multiple purposes and usually proves to be a good investment (Argent et al. 2016; Merritt et al. 2017). In addition to draughting the initial modelling structure, conceptual modelling provides modellers and endusers with a means of visualising and discussing the land-water system and summarising the current state of knowledge. It also helps modellers explain or reiterate model assumptions, knowledge gaps and sources of uncertainty. And it allows endusers to confirm that their objectives, constraints and prospective management actions have been incorporated in the modelling process and are documented accurately.

Developing Spatial Frameworks

Compatibility among component models, including compatibility in the spatial and temporal resolution of data exchanged among the models, is a basic requirement for LWS modelling (Laniak et al. 2013). Achieving this compatibility is one of the challenges of converting descriptive conceptual models into quantitative models. A useful way to approach this challenge is to focus on establishing the spatial framework on which each of the component models will operate as a first step in quantitative model selection and development. The spatial framework serves to define all spatial entities that are indicated or implied by the conceptual model, provide a structure to contain data describing each spatial entity that will be used as model input and codify how the entities and component models are connected.

Developing a spatial framework requires the modelling team to define the spatial entities that are defined by the conceptual model (either explicitly or implicitly) at an appropriate level of resolution (Greene et al. 2015; Elliott et al. 2016). For example, any component model that is representing catchment processes, such as contaminant transport and attenuation, must at least resolve the sub-catchments upstream of each receiving environment. At a minimum, the spatial framework must define aquatic receiving environments as whole entities. However, depending on how ecological responses in these environments are represented by the conceptual model, they may need to be further subdivided (e.g. using two- or three-dimensional grids; Mooij et al. 2010). Therefore, the development of the spatial framework is not independent of the subsequent model selection process, and these two processes often need to be iterated.

As well as defining the spatial entities represented by the model, the spatial framework must provide a basis for defining their interconnections so that spatially compatible information can be passed between component models. For most LWS models, connectivity between spatial entities is represented by surface and subsurface hydrological flowpaths (e.g. Wei et al. 2019). In some region-scale LWS models, spatial entities are connected by both atmospheric and hydrological transport pathways (e.g. Hood et al. 2021). In all cases, the transport pathways need to be represented in the spatial framework.

We recommend developing the spatial framework at the start of the formal modelling stage to help the modelling team understand input data requirements for component models, and how compatibility between these components will be achieved. It is useful if all spatial entities, their associated data, and their linkages are held in a GIS, which provides a means for spatial data input and data exchange between models, and for storing and visualising model output. Several widely used hydrology, water quality and aquatic ecosystem models have GIS interfaces to manage data transfer between spatial frameworks and component models (e.g. Winchell et al. 2013).

Historically, the development of spatial frameworks for modelling has been impeded by shortages of the data needed to characterise drainage networks, receiving environments, land and land use (e.g. Bartley et al. 2012). These data shortages have been progressively alleviated through advances in remote sensing and other environmental monitoring technologies, and the provision of open-access spatial datasets (e.g. Jakovljević et al. 2019; Radočaj et al. 2020; Hermosilla et al. 2022).

Integrating Biophysical and Economic Models

Decisions about land management actions may require a combination of biophysical and economic modelling (Roebeling et al. 2009; Schlueter et al. 2012). In some cases, the biophysical and economic modelling tasks are carried out separately and the output from each task is evaluated independently (Bosch et al. 2018). In other cases, biophysical and economic component models are integrated, particularly when the modelling objective is some form of optimisation, such as achieving a water quality objective with minimal cost (Doole et al. 2013; Beverly et al. 2016; Liu et al. 2019, 2020).
Assessments of economic impacts of land management actions may need to consider a range of scales from the enterprise scale (either individual land-use enterprises or classes of enterprise) to flow-on impacts that must be assessed at the catchment, regional or national scale (e.g. Bryan et al. 2009). Flow-on impacts are changes in economic activity that occur outside of individual enterprises and that accumulate in space and propagate to the wider economy of the geographical area being considered. The wider economy includes monetary transactions between enterprises, between enterprises and the government, and between households and enterprises, and imports and exports. In our conceptualisation of LWS models, economic impacts of land management actions are assessed as both enterprise-level impacts and flow-on impacts at larger scales (Fig. 1).

Model-based assessments of economic impacts at the enterprise-scale require financial models of enterprises and are likely to be a minimum information requirement for regulatory processes; this information is needed by decision makers to judge the fairness and equity of proposed management actions to land users. Typically, indicators of economic impacts at the enterprise-level are measures of profitability that may be expressed as averages and ranges or distributions. Modelling flow-on economic impacts involves aggregating enterprise level impacts over larger spatial scales and representing the links between enterprises and the wider economy. Indicators of flow-on impact concern economic welfare in the wider economy (e.g. regional employment levels). Because additional processes are being represented, predictions of flow-on economic impacts are inevitably more uncertain than predictions of enterprise-level economic impacts.

Ideally, individual enterprises are represented in the spatial framework and in the corresponding LWS model, by the land areas they occupy (e.g. individual farms). When this is not possible due to data shortages, it may be sufficient to represent different classes of enterprises as combinations of specific classes of land use (e.g. forestry, pastoral farming, cropping) and environmental characteristics (e.g. climate, topography, soil types) (Greene et al. 2015). Whether they are defined by individual enterprises or classes of land use enterprises, the corresponding land areas are the basic spatial units for both economic and biophysical model chains. LWS simulations are initiated by specifying changes to land areas in the form of combinations of land and water use choices and management actions (Fig. 1). Sets of land area specifications comprise input data for the economic and biophysical model chains, and the outputs from both model chains is then paired by land area. This pairing has the benefit of providing endusers with information that describes the trade-offs between economic costs and environmental protection that are needed to make normative decisions.

We recommend that modelling enterprise-level economic impacts is prioritised over the flow-on economic impacts over larger scales, and that the need for modelling the flow-on economic impacts, and the scale over which these are assessed, is carefully considered given the increased complexity and uncertainty of these predictions. Modelling of flow-on economic impacts should be undertaken when the scope of the regulatory changes being considered are likely to significantly impact the wider economy, and the size and character of these impacts are relevant information that is needed by the decision-maker. Determination of whether the preceding criteria are met is a judgement that should be informed by deliberations at problem framing and conceptual modelling stage of the LWS model development.

We note that numerous LWS projects that have omitted economic models would likely benefit from their inclusion, particularly those projects that consider land management scenarios with potentially large socio-economic impacts. Examples include several LWS models of lakes and rivers and their catchments (Guse et al. 2015; Zia et al. 2016; Bucak et al. 2018; Motew et al. 2019). In each of these cases, scenarios involving extensive changes in agricultural and/or urban land use were simulated. While economic modelling may have been out of scope for these projects, information about trade-offs between the environmental and economic effects of the scenarios is likely to be needed to inform land management decisions concerning each of the modelled systems.

The Chesapeake Bay Project modelling system provides a useful example of the integration of biophysical and economic component models. For most of the project's history, comparisons of effectiveness of land management actions and their economic costs were made outside of the modelling system, using simple spreadsheet tools. However, a land management optimisation tool has now been integrated with the catchment water quality model for the purpose of identifying the most cost-effective land management actions in terms of contaminant load reductions and capital and operating costs, at any location in the catchment, subject to constraints such as the land area available for constructing contaminant retention systems, and the minimum allowable load reduction (Kaufman et al. 2021). This tool has helped the project modellers to identify substantial cost savings for agricultural and urban land management actions compared to the land management plans prepared by local and state governments, for a given level of contaminant reduction.

Model Selection and Coupling

Our first recommendation regarding model selection is to reuse a selection of extant, open-source models for multiple projects, to the degree possible. Using extant models as components in LWS models is generally faster and less costly than developing new component models, and benefits from greater familiarity among modellers and endusers with a selected group of well-tested models (Robinson et al. 2004; Holzworth et al. 2010). These advantages apply to two types of model reuse: repurposing models in different combinations to produce new types of output, and transferring model applications from the environmental settings they were originally designed to simulate to new settings. Model repurposing and model transfer may require reparameterization and recalibration and may also require structural modifications of the extant models. However, a substantial level of modification would be required to make development of one or more new models the more efficient choice. In addition to the time and cost savings, many extant models have track records of testing and peer review, which provide an assurance of model reliability (Jones et al. 2020). Increasing efficiency through model reuse has been recommended repeatedly (e.g. Robinson et al. 2004; Holzworth et al. 2010, McIntire et al. 2022). Despite these reminders, new environmental models have proliferated with substantial duplication of effort, as indicated by recent reviews and inventories (e.g. Wang et al. 2013; Janssen et al. 2015). In the specific case of LWS modelling, new models intended for use in assessing land management scenarios are developed and published regularly (e.g. Yang et al. 2018; Hankin et al. 2019). It is not clear from these publications whether the new models were required to meet stakeholder needs, or if extant model reuse was considered before commencing with the development of new models.

Some extant models that are potentially useful for LWS modelling are proprietary, with restrictive user licences. In unmodified form, these models may be incompatible with other models, and making the necessary modifications may violate license provisions. When modifications are permitted, they may require specialist programming skills. These impediments can be reduced by using open-source models (Morin et al. 2012). In addition, open-source models benefit from tests and improvements made by the modelling community (Bruce et al. 2018; Fu et al. 2020). While excluding proprietary models reduces the choice of component models and model combinations, several recent LWS modelling projects have used combinations of open-source models exclusively (e.g. Shabani et al. 2017; Bucak et al. 2018).

Specific project objectives also act as filters to reduce the choice of suitable models. However, the number and diversity of open-source models is increasing, including open-source versions of proprietary models (Yuan et al. 2020; Soares et al. 2021). In cases where no suitable open-source models exist, the modelling team must choose between developing new models, substantially modifying extant models, or negotiating with the licensors of proprietary models.

Our second recommendation is to assemble component models that adequately represent the land water system components and processes defined by the conceptual model. Here model adequacy means that an LWS model simulates the land-water system with sufficient detail, accuracy and precision to meet enduser needs, within the constraints discussed above (e.g. limited time, funding, data). This view of model adequacy equates to reaching an acceptable balance between reliability and feasibility (Getz et al. 2018). Increasing reliability may entail increasing model complexity by increasing the numbers of parameters and processes. Increasing feasibility generally entails decreasing model complexity by removing or aggregating parameters and processes. Highly complex models have high data requirements, are challenging to parameterise and may have long run times, which consumes time and funding, whereas very simple models may miss or inadequately represent key processes (Getz et al. 2018; Hamilton et al. 2022). If candidate models are judged to be overly complex or overly simple, they can be systematically simplified or elaborated, respectively (Moriasi et al. 2015, Hong et al. 2017; Getz et al. 2018). Evaluations of model adequacy are subjective and must be made in the context of the land-water management problem, with consideration of issues such as tolerance of uncertainty, data availability, and time and funding constraints. As noted above, transparency is a fundamental criterion for environmental modelling and this applies to evaluations of model adequacy for a given problem.

The component models in integrated modelling projects should be comparable in complexity and detail, and modellers should avoid over-investing in one component model where the detail (e.g. spatial and/or temporal resolution) is not utilised in the other models in the chain. For example, considerable effort may be required to model contaminant fluxes at a daily timestep, but there is little benefit if the purpose is to provide input to an ecological model that operates at a seasonal or annual time step.

Model coupling refers to data transfer between the component models in a model chain, and the processing steps that ensure the output data from one model are appropriately structured as input to the next model. Coupling may also include more intensive model integration, including runtime coordination and feedback between models through dynamic, two-way data exchange.

The terms ‘loose coupling’ and ‘tight coupling’ are often used to characterise levels of model integration and data exchange (Brandmeyer and Karimi 2000). Loose coupling refers to integration with minimal modification of the component models. Data exchange between loosely coupled models is generally limited to one-way transfer, with no feedback. The data transfer may be automated using interface software or web services, or manual. Manual coupling (i.e., extracting and reformatting output data from one model and uploading the data to a second model, which is then executed in isolation of the first model) is the simplest type of loose coupling (e.g. Kim et al. 2008). Manual coupling requires no modification of the component models and no external software. While the initial costs for manual coupling may be low, these costs increase as the number of component models and/or the number of simulations increases.

Tight coupling refers to high levels of model integration, including dynamic feedback (e.g. Yalew et al. 2018; Du et al. 2020). Tightly coupled models are usually managed by external software frameworks (Belete et al. 2017; Buahin and Horsburgh 2018; Harpham et al. 2019). To use these frameworks, the component models must be compliant in terms of runtime control, data and file formats and other properties. Noncompliant models require modification or software ‘wrappers’ that translate between framework and model functions (Johnston et al. 2017). Most of the frameworks in current use were developed for models from specific domains such as hydrology and soil science, which may be unfavourable for LWS modelling projects that need to link models from multiple domains (e.g. hydrological, water quality, hydrodynamic ecological models). Furthermore, the high level of software expertise required to modify component models and/or operate frameworks, and the additional costs and time required can make tightly coupled models unfeasible.

When using coupled models to represent land-water systems, the need for dynamic feedback between models is limited by the predominately unidirectional movement of water and contaminants. For example, LWS models rarely need to represent feedback from drainage networks back to contaminant source areas on land, or from receiving environments back to drainage networks. For this reason, unidirectional data transfer between loosely coupled models is sufficient for many LWS modelling applications (e.g. Migliaccio et al., 2007, Debele et al. 2008; Shabani et al. 2017). Possible exceptions include the simulation of bidirectional hydrologic exchange between surface water, groundwater and the atmosphere (e.g. Tian et al. 2012, Guzman et al. 2015). In the case of the Chesapeake Bay Project, all major component models (e.g. water quality, air quality and estuarine ecosystem models) are loosely coupled to facilitate the use of these models independently (Hood et al. 2021). However, some routines within the component models are tightly coupled where dynamic feedback is required (e.g. between estuary biogeochemistry, primary production and hypoxia parameters).

Our recommendation concerning model coupling is to use loosely coupled models unless the conceptual model has explicitly identified processes that can only be represented by dynamic feedback between models. Loose coupling is faster and requires less programming expertise than tight coupling, and maximises the range of suitable, extant models. Further, the need for flexibility (e.g. model recombination) in LWS modelling to address a range of land management problems, and the time and funding limits facing regulatory agencies and stakeholders both favour loose-coupling as a default approach. We also recommend comparing the costs of manual coupling versus the use of interface software and web services for data transfer. It is likely that a one-time investment to automate data transfer will be less costly than manual coupling when running many simulations. However, we note that some interface software and web services are at early stages of development, and their use may require modelling teams to expand their range of skills (Gregory et al. 2020; Tucker et al. 2022). As an alternative to both manual coupling and reusable interfaces, modelling teams may develop simple protocols for data transfer for each LWS project (Gregory et al. 2020).

Conclusion

Land-water management problems are vexing for regulatory agencies, stakeholders and decision-makers. These groups are tasked with identifying land management actions that will reduce the adverse effects of land use on aquatic environments while minimising collateral effects on socio-economic values. Judgements about the most appropriate management actions are inevitably normative ones, but they should be well informed. The essential information needs are options in the form of alternative land management actions, and assessments of the effectiveness of those actions and their economic impacts. Identifying the most effective management actions after the fact (i.e., by trial and observation) is not practical due to the long time lags between the implementation of management actions on land and the detection of responses in receiving environments (Jeppesen et al. 2005; Vervloet et al. 2018). Instead, the effectiveness of alternative management actions needs to be predicted in advance, and the selection of actions to be implemented needs to be based on those predictions.

We argue that LWS modelling using coupled biophysical and economic models is a rigorous and transparent approach for predicting and comparing the effects of alternative management actions. LWS modelling may be more time-consuming and costly than less rigorous prediction methods such as expert opinion, but it may prove less costly in the long run, by helping decision-makers reduce the risk that management actions fail to achieve their intended objectives. Furthermore, LWS modelling can be made far more time- and cost efficient by shifting the general approach away from bespoke model development, towards an integrated modelling approach that maximises model reuse and transferability.

For LWS modellers participating in regulatory processes, the true measure of success is model uptake, i.e., acceptance of model output by endusers and use of that output in decision making. To improve the likelihood of uptake, modellers must first ensure that their work is deemed fit-for-purpose by endusers. Fitness-for-purpose is a subjective quality that entails an acceptable balance of model reliability, feasibility and usefulness, plus transparency. The first three criteria are subject to trade-offs, but transparency is a fundamental requirement that should not be traded off.

The principal contributions of this paper are the recommendations set out in Section 5. In contrast to the wealth of technical and behavioural advice produced for the broader environmental modelling community, our recommendations are aimed specifically at multi-disciplinary modelling teams that are commissioned to develop and apply LWS models as part of environmental regulatory processes. Some of our recommendations are consistent with feedback from surveys of modellers and model endusers (Merritt et al. 2017; Will et al. 2021), but other recommendations contradict the advice and examples of previous authors, including public involvement in all stages of model development and application, reliance on generalist facilitators, and the development of complex, bespoke models and tightly coupled models when extant, loosely coupled models are suitable. The wide-ranging recommendations in Section 5 are linked by the proposition that success in LWS modelling projects is a matter of meeting the needs of stakeholders and decision makers with fit-for-purpose models, not creating highly advanced or novel models per se. To succeed in these terms, modellers need to view themselves as honest brokers and problem solvers rather than detached, ivory-tower scientists, and view their models as means to assist decision making rather than ends in themselves. We note that the recommendations set out in Section 5 are not limited to LWS modelling; they applicable to other fields where modelling teams use integrated environmental models to organise multidisciplinary knowledge and data, carry out scenario assessments and explain the results of those assessments to participants in the regulatory process. These fields range from water resources management, to solid waste management, to renewable energy development (Levis et al. 2013; Gils et al. 2017; Hülsmann et al. 2019).

Increasing LWS model uptake is strongly dependent on the mix of skills provided by the modellers. In this paper, we have made the case that modellers who are effective contributors to environmental regulatory processes have a particular mixture of technical and non-technical skills. In addition to technical expertise in model development, these modellers are effective knowledge brokers and have problem-solving mindsets. Unfortunately, this mixture is rare and the shortage of modellers with the requisite skills and aptitudes is a limiting factor for LWS model uptake. Alleviating the shortage requires investing in early-career environmental modellers and providing them with opportunities to gain experience. Similar appeals have been made to invest in training ‘translational ecologists’, ‘environmental implementation scientists’ and ‘boundary spanners’, all of whom work directly with environmental decision makers (Schwartz et al. 2017; Cooke et al. 2020). As is the case for these non-traditional science roles, conventional academic programmes may be poorly suited for training LWS modellers; regulatory agencies and applied research institutions and consultancies are better placed to provide real-world experience.

LWS models are powerful tools, as indicated by cases where land management actions have been implemented based on model output and the outcomes of those actions were consistent with model predictions (e.g. Hood et al. 2021). However, need for LWS modelling to inform land management decisions is certain to exceed the capacity of the environmental modelling community unless three things happen: (1) more time- and cost-efficient modelling approaches are developed and adopted; (2) the modelling workforce is expanded; and (3) more modellers make the shift from a model-centred perspective that emphasises model complexity and technical performance to a problem solving perspective that emphasises the acceptance and use of model output by endusers.

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