

DECISION POINT

*Connecting conservation policy makers,
researchers and practitioners*

Issue #107 / December 2018



Our final bumper issue: 9 lessons in good environmental decision making

1. The field of decision analysis, 2. building models,
3. prioritising environmental investments,
4. valuing information, 5. adaptive management,
6. long-term monitoring, 7. influencing policy,
8. marine collaboration & 9. social values.



Inside this issue

Many legacies	3
Navigating the field of decision analysis	4
Building models for better decisions	12
Prioritising environmental investments	20
Valuing information for better decisions	26
Learning about adaptive management	34
The importance of long-term monitoring	40
Decision science and environmental policy....	46
Collaboration across the sea	52
Social dimensions of good decision making...	58



Decision Point is the free research magazine of the Centre of Excellence for Environmental Decisions (CEED). CEED is a network of conservation researchers working on the science of effective decision making to better conserve biodiversity. Our members are largely based at the University of Queensland, the Australian National University, the University of Melbourne, the University of Western Australia and RMIT University.

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On the point

A good decision for the environment

What's a good decision for the environment? Is it: More funding for biodiversity conservation? a new national park being declared? a threatened species being declared a state emblem to give it a higher profile? or a program to pay farmers for good environmental stewardship?

All of these proposals would be sold by the government (or agency or NGO) implementing them as good decisions for the environment. But are they? Even if they generated the outcomes that were promised, is that enough?

As a younger man I believed a good environmental decision was one that generated a good outcome (saved a species over here, protected valuable habitat over there). While I still think a good outcome is important, it is possibly secondary to the process by which it is generated. This I believe is the key insight I have taken away from being associated with the Centre of Excellence for Environmental Decisions for the past seven years: good decision making is more about the process of making decisions than the direct outcome of any particular decision.

A good decision for the environment is one that is transparent, efficient and effective; that came about with real stakeholder engagement and support; that enables learning; and something that serves as a stepping stone to even better decisions down the line.

CEED has been working on the science behind every dimension of good decision making. It has been my privilege to bring you stories in *Decision Point* from this body of research. Along the way, I'd like to think we've helped build a community of interest in environmental decision science and good decision making.

CEED didn't invent decision science but, through its research and tool development, it has played a pivotal role in placing environmental decision science at the centre of good conservation management and policy. To celebrate that achievement, this final issue of *Decision Point* is a bumper 64-page issue carrying nine stories on the dimensions of good decision making.

Six of these stories have appeared in earlier issues of *Decision Point*, though the versions in this issue are expanded with case studies taken from across the life of CEED. And there are three additional stories (on value of information, collaboration and social dimensions) that were created specifically for this issue.

I'd like to thank all the excellent people I've worked with over the years, too many to list here, with special mention of Kerrie Wilson and Hugh Possingham (CEED's two Directors) for their enthusiastic support of *Decision Point*. And I'd like to acknowledge the fabulous and supportive feedback we've had from our readers over the years.

All the best

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DECISION POINT #107

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Many legacies

And one final bumper issue of *Decision Point*

By Kerrie Wilson (Director, CEED)

All good things come to an end. The challenge is to learn from our investments (our choices and decisions), savour the good bits, learn from the not-so-good bits, and move on.

As 2018 closes, so does CEED. We've come to the end of our funding but we leave a legacy that I hope will serve Australia and the global conservation effort for generations to come.

The vision that we set forth for CEED was to become the world's leading research centre for solving environmental management problems and for evaluating the outcomes of environmental actions. As CEED reaches the end of its funding cycle, how have we fared?

In terms of outputs, CEED has been a stellar performer at producing scientific publications and delivering cutting-edge research. Our science is widely cited by researchers around the globe, but also in primary government reference materials such as the State of the Environment Report. Indeed, CEED's contribution to the environmental sciences has made a major contribution to Australia's high standing in this field.

In our final years we initiated an effort to evaluate the impact of our Centre of Excellence. We now have evidence that CEED has achieved more impact in environmental decision making than would have been possible by the sum of all the individual, smaller research groups that makes up CEED. This is due to the networks and collaborations that have been possible as a result of the significant investment from the Australian government (in the Centre of Excellence program). As such, our evaluation also captures how CEED has contributed to training the next generation of research leaders and improving research communication and outreach.

Of course, one of CEED's greatest legacies will be its people. CEED commenced in 2011 and has now generated a well-connected alumni network that spans the globe. These incredibly capable researchers are receiving national and international recognition for their achievements. They are securing coveted positions in the academy, and taking up leadership roles in governmental and non-governmental organisations.

Over the life of our Centre, the field of environmental decision science has grown from a little utilised academic pursuit to become a cornerstone of environmental policy. Our sustained efforts at research outreach and communication has been fundamental for this achievement; which brings me to this issue of *Decision Point*.

Decision Point is a fine example of CEED's communication and outreach. This final issue is #107! It's been running for over a decade (with CEED as the major supporter), generated well over 1,000 research stories, and we've also produced four issues of a Spanish-language version (*Decision Point en Español*). These are considerable milestones, but it also represents a significant investment made by CEED in communicating our research in an attractive, engaging and ongoing manner for the benefit of the wider community.

The feedback we get on *Decision Point* has been uniformly praising and, along with our other communication and outreach activities, *Decision Point* has made a significant



One of CEED's legacies will be our attempt to develop a framework to measure CEED's impact, a methodology that can be applied to any research network (see [Decision Point #106](#) for details). Pictured here is the working group that developed the framework. From left (front row) are Sarah Bekessy, Dave Pannell (who led the evaluation), Tas Thamo, Tammie Harold, Eve Macdonald-Madden; (middle row), Kerrie Wilson, Jonathan Rhodes, Richard Hobbs, Margaret Byrne; (back row), Tony Peacock, Alistar Robertson, Vanessa Adams and Kathy Avent.

contribution to a cultural shift in policy formulation in Australian governments at all levels; a shift away from ad hoc, opaque decision making (in regards to policy relating to nature, environment and threatened species management) towards more accountable, systematic and adaptive decision-making.

And this final issue is indeed a bumper issue consisting of nine extended stories on different dimensions of environmental decision science (all authored by CEED CIs; the idea for this collection began at a writing retreat at the beginning of 2017). Each story provides several case studies from the history of CEED (and *Decision Point*) so this issue makes for a nice reflection on the life and times of CEED – which, in retrospect, has been a fabulous time full of rich learnings, valued comradeship and important discoveries.



CEED CIs and support staff at a writing retreat last year - the genesis of this bumper issue of *Decision Point*. Pictured from the left are Sarah Bekessy, Peter Vesk, Salit Kark, Kerrie Wilson, Michael Bode, Jonathan Rhodes, Mike Runge, Dave Pannell, Eve Macdonald-Madden, Kathy Avent, Brendan Wintle, David Lindenmayer, Terry Walshe, Iadine Chades and David Salt.



Navigating the field of decision analysis

Helping decision makers frame, analyze, and implement decisions

By Michael C. Runge (U.S. Geological Survey) and Eve McDonald-Madden (University of Queensland)

Managers, policy makers, and decision makers with responsibility for environmental decisions have an extraordinarily difficult job. The systems they manage are complex (coupled human-natural systems), with many dimensions and complicated dynamics. Our knowledge of how those systems respond to management actions is often limited, so many of the decisions have to be made in the face of uncertainty.

Key messages:

1. All decisions have the same recognizable elements.

Context, objectives, alternatives, consequences and deliberation. Decision makers and analysts familiar with these elements can quickly see the underlying structure of a decision.

2. There are only a small number of classes of decisions.

These classes differ in the cognitive and scientific challenge they present to the decision maker; the ability to recognize the class of decision leads a decision maker to tools to aid in the analysis.

3. Sometimes it is worth collecting more information, sometimes it isn't. The role of science in a decision-making process is to provide the predictions that link the alternative actions to the desired outcomes. Investing in more science is only valuable to a decision maker if it helps to choose a better action.

4. Implementation. The successful integration of decision analysis into environmental decisions requires careful attention to the decision, the people, and the institutions involved.

Above: The field of environmental decision analysis has come a long way in recent decades. So much so that the richness of decision-making approaches can sometimes seem overwhelming. Underlying that richness are a few basic elements and processes. In this article we present some key elements to help point the way.
(Photo by David Salt)

The field of decision analysis provides a comprehensive set of tools for structuring, analyzing, and making decisions. Arising initially as a way to understand and manage risk, the field has expanded in the last 80 years to cover such topics as multiple-objective trade-offs, time-dependent linked decisions, the value of information, and competition among multiple decision makers. At the same time, cognitive psychologists have studied human decisions to understand when our innate processes work and when they fail.

Formal decision analysis is increasingly being used in many sectors, including economic, industrial, manufacturing, agricultural, transportation, and medical sectors, by individuals, corporations, non-profits, and government agencies. Occasional application of decision analysis to environmental decisions began in the mid-1970s, but the environmental science and management world was largely unaware of decision analysis until the late 1990s and early 2000s. (A major aim of the ARC Centre of Excellence for Environmental Decisions – CEED – was to better connect decision analysis and environmental management).

Today, the field of environmental decision analysis is coming of age with the maturation of a rich set of tools to help decision makers frame, analyze, and implement decisions. Sometimes the wealth of tools is daunting, but there are core principles that tie decision analysis together. To help the reader navigate this complexity, we summarize the field of decision analysis in four key messages.

1. All decisions have the same recognizable elements

The decisions we have to make are normally surrounded by a fog of complexity. One of the most valuable contributions of the field of decision analysis is the recognition that all decisions have a common set of elements: context, objectives, alternatives, consequences, and deliberation. These elements form a useful guide for the decision maker.

Context: Albert Einstein once said that if he had 1 hour to save the world he would spend 55 minutes defining the problem and 5 minutes finding the solution. Clarity about the decision context leads to an efficient search for solutions.

Understanding the decision context begins with some simple questions (simple to ask, anyway). What is the problem that is being faced and what triggered it? Who will make the decision and what authority do they have to act? Who else cares about and can influence the decision? What is the geographic and temporal scope of the problem? What ecological, social, and legal background is relevant?

Objectives: The decisions that we make are driven not by science but by the values we hold. Understanding the value sets of the decision makers, the organisations responsible for decisions, and the key stakeholders is a crucial step in a structured approach to decision-making.

The values at the heart of any decision are the fundamental objectives—the long-term outcomes we are trying to achieve by implementing a course of action. There are often many of these objectives, and they might conflict. Thus, we might care about the long-term conservation of wetland birds at the same time we care about maximizing agricultural production and minimizing flooding risk.

We often lose track of fundamental objectives by focusing instead on means objectives. These are potential ways by which we might achieve our fundamental objectives. For example, we might focus on target levels for irrigation withdrawal and argue over those, forgetting that what we really care about are the birds, the crops, and the flood protection.

Alternatives: Once we know what we want to achieve with a decision, it's possible to start thinking about what we could

Decisions about willows in the high country

Here's an example of how structured decision making (SDM) was used to help develop a long-term management strategy for the invasive gray sallow willow up on the Bogong High Plains of Victoria (Moore & Runge, 2012). The stakes were high with the prospect of this very invasive willow taking over an endangered alpine ecosystem. There is great uncertainty surrounding the available options and, looking into the future, this is compounded by a changing climate and shifting fire regimes.

Structured decision making involves working with key stakeholders involved in a problem to create an agreed framework around the decisions they need to make. The process used involved setting a context, agreeing on objectives, listing the various available options to meet these objectives and devising ways to compare the costs and benefits of those options.

During a three-day structured decision making workshop it emerged during this time that the major decision faced by the willow managers was where to focus their control effort. The removal of willows from EPBC listed alpine bogs was identified as the main objective of management so the control of willows in bogs seemed a good place to start.

However, willow establishment is facilitated by wildfires which are predicted to increase with climate change. Should some effort be allocated to removing mature seed-producing willows in nearby waterways to prevent invasions in the future? If so, how much effort? In the absence of a strategy, managers had been allocating effort to bogs and waterways guided by a combination of intuition and available resources.

The scientists used structured decision making to formally describe this decision. They then built a stochastic model of the spread of willow onto the Bogong High Plains and used it to calculate how different management strategies would affect the amount of willow in bogs over the next 200 years. The model contained approximately 40 factors or parameters; most of these had never been measured and were highly uncertain.

They incorporated this uncertainty into the model by choosing parameters from probability distributions (that represented our uncertainty about the parameter values) and re-running the model 10,000 times. Using the model, they identified where control effort should be focused to minimise the abundance of willows in bogs over the longer term. This was done by calculating which management alternative worked best on average across the 10,000 possible scenarios.

The optimal strategy was to allocate all available effort to the bogs until the budget exceeded 2,000 work days per year. It needs to be noted that 2,000 work days per year is four times the current budget levels. Beyond this point it was optimal to allocate some effort (20-60% total budget) to eliminate populations of seed-producing willows in nearby rivers. Effort was allocated to the closest populations first and then to more distant waterways as the budget increased.

Reference

Moore JL & MC Runge (2012). Combining Structured Decision Making and Value-of-Information Analyses to Identify Robust Management Strategies. *Conservation Biology* 2: 810–820. And see the story on this paper in [Decision Point #67](#)



A Parks Victoria employee struggling to see the bog for the willows. (Photo by Carmen de Rooze)



Structured decision making involves working with key stakeholders involved in a problem to create an agreed framework around the decisions they need to make. Pictured here is Mike Runge discussing models and uncertainties with managers tasked with dealing with invasive willows in Victoria's highlands. (See the box on decisions about willows in the high country.)

do to achieve these outcomes. Unfortunately, most people reverse this process, assessing what they could do before understanding why they are doing it.

While working out what actions we can take to achieve our objectives seems simple, the process is hampered by psychological biases. Often we anchor on the decisions we have made in the past or small variations on those decisions (see [Decision Point #93](#)). But to achieve a good decision, we need to be creative, to think beyond our current set of actions and our means objectives, and instead focus on developing actions that may help us achieve our fundamental objectives and expose trade-offs within these objectives. Indeed, given the challenges we face in environmental management, creative solutions are needed more than ever, and sticking with business as usual is unlikely to solve the problems we have.

A creative search for alternative actions should challenge constraints. Many constraints to a decision are only perceived, not real. We may be told, for example, that a decision has a firm budget constraint, but creative consideration of leveraged funding might open up new alternatives.

Prediction of consequences: The role of science in decision making is to make predictions. There are no decisions without prediction. If you decide what bus to take, you think about what time you need to get somewhere and predict which bus will get you there in time. We make predictions daily for every decision we make.

What are these predictions? They are a mapping of the actions we are choosing among (the alternatives) to the things we want to achieve (the objectives). The challenge for the scientist is to use the best available information to make these predictions, while comprehensively accounting for uncertainty.

Selection of preferred alternatives: By this point of the process we have evaluated each action or set of actions against each objective. But no decision has been made, yet, and indeed, the best choice might not be obvious. In the deliberations to find a preferred alternative, the decision maker might need to weigh the multiple objectives to find the right balance, or consider the risk posed by uncertainty, or consider how this decision will affect future decisions. These are challenges at the interface of science and values, and the field of decision analysis provides a large set of tools to help.

2. There are only a small number of classes of decisions

Many decisions resemble each other, and this recognition is one of the ways that people become better decision makers and analysts, because they realize they have made a similar decision in the past. For a decision analyst, each class of decisions has a common structure, poses a similar challenge to the decision maker, and benefits from a similar set of analytical tools. Although the circumstances and details of each environmental decision are unique, recognizing the class of decision accelerates the process of analysis. Here we discuss five classes of decisions that arise commonly in environmental contexts.

1. Risk analysis: Coupled socio-ecological systems are complex, our knowledge of how they respond to management actions is limited, and some aspects of their dynamics are out of our control. Thus, many environmental decisions are made in the face of uncertainty and are open to risk. A grassland manager, for example, in an effort to hold back succession, can choose between mowing and burning. The effects of mowing are better understood and more controllable, but the disturbance does not return the same nutrients to the soil nor does it induce germination. On the other hand, even with careful preparation, it is hard to fully control a burn. What are the risk trade-offs in choosing among these actions?

Risk decisions are ones in which the precise outcomes of the alternative actions are not known prior to implementation. The challenge for the scientist is to estimate the probabilities of the various outcomes under the different alternatives. The challenge for the decision makers is to articulate their degree of risk tolerance. The methods for risk analysis are well developed and have been applied in a number of environmental contexts, including endangered-species management, biosecurity, and pesticide regulation.

2. Project prioritisation: Consider an agency that has \$300,000 to spend to restore instream, riparian, and upland habitat for the benefit of a suite of threatened endemic species. The agency has received several dozen proposed projects, ranging in cost from \$10,000 to \$175,000. Taken together the projects cost \$1.4 million, far in excess of the available funds. Which of the projects should be funded to assure the most benefit for the species in question?

This common class of problem goes by many names including portfolio allocation, the knapsack problem, or combinatorial optimization. In the environmental world it is frequently called project prioritization (see [Decision Point #29](#), p8-10; and [Decision Point #103](#)). The challenge is to select a subset of a large number of possible projects, subject to a budget or resource constraint, that maximizes the combined benefit.

The number of possible portfolios to select is typically very large (for example, with 24 projects to select from, the possible number of combinations is $2^{24} = 16.8$ million). The benefit of adding a particular project may depend on what other projects are in the portfolio, because there could be synergies, conflicts, or redundancies among projects. The analytical challenges are to calculate the net benefit of a candidate set of projects, then search among the large set of possible portfolios for the optimal one.

Note that a broad set of decisions is included in this class. The 'projects' could be species to save, invasive species to combat, research projects to fund, land parcels to acquire, varieties of actions to implement, etc. The resource constraints could be

money, staff time, the total number of projects to undertake, or even the number of individuals of a threatened species available for translocation.

3. Spatial planning: Spatial planning is a special case of project prioritisation. The individual 'projects' are spatial elements (eg, land parcels, grid squares), and the challenge is to identify the best set of spatial units to include in the reserve to maximize some measure of environmental benefit, while either staying within budget or minimizing the area required. The nature of the interactions among the units can be important, reflecting the benefits of connectivity but possibly also the value of spatial independence.

Decision analysis and monitoring

Decision analysis is a procedure for discriminating between suitable courses of action; in the case of monitoring, it helps to select the most appropriate regime for monitoring (see Fig. 1: Decision 6, Question 13) or management (see Fig. 1: Decisions 10, 14, 15). Decision analysis involves a structured enquiry into the different options available to manage or monitor, along with their costs, benefits and constraints. A simple form of decision analysis ranks options according to their expected cost effectiveness, or expected benefit divided by expected cost.

When deciding which **management option** is best, one must consider the benefit of each possible action in terms of reaching the overall program objective, the probability of success of that action, and of course the cost of implementing that action.

To select the best **monitoring regime** we include the same components, but the benefits now relate to the quality of information needed to make a decision based on the reason for monitoring (eg, track system state to guide state-dependent management, or track performance to guide adaptive management). Furthermore, adequate monitoring must consider the ability of the strategy to detect changes in the system.

Acquiring **information** on benefits and costs for a decision analysis can be achieved through expert elicitation or through more detailed scenario modelling. Options for implementing decision analysis range from a simple calculation of the combined benefits relative to the total costs

incurred (eg, Benefit x Probability of success / Cost) to a more complex optimisation (eg, stochastic dynamic programming or reinforcement learning). Methods of obtaining data and implementing decision analysis vary in their cost and their ability to provide rigorous results.

Reference

McDonald-Madden E, PWJ Baxter, RA Fuller, TG Martin, ET Game, J Montambault & HP Possingham (2010). Monitoring does not always count. *Trends in Ecology and Evolution* 25: 547-550. And see [Decision Point #52](#), p4-6.

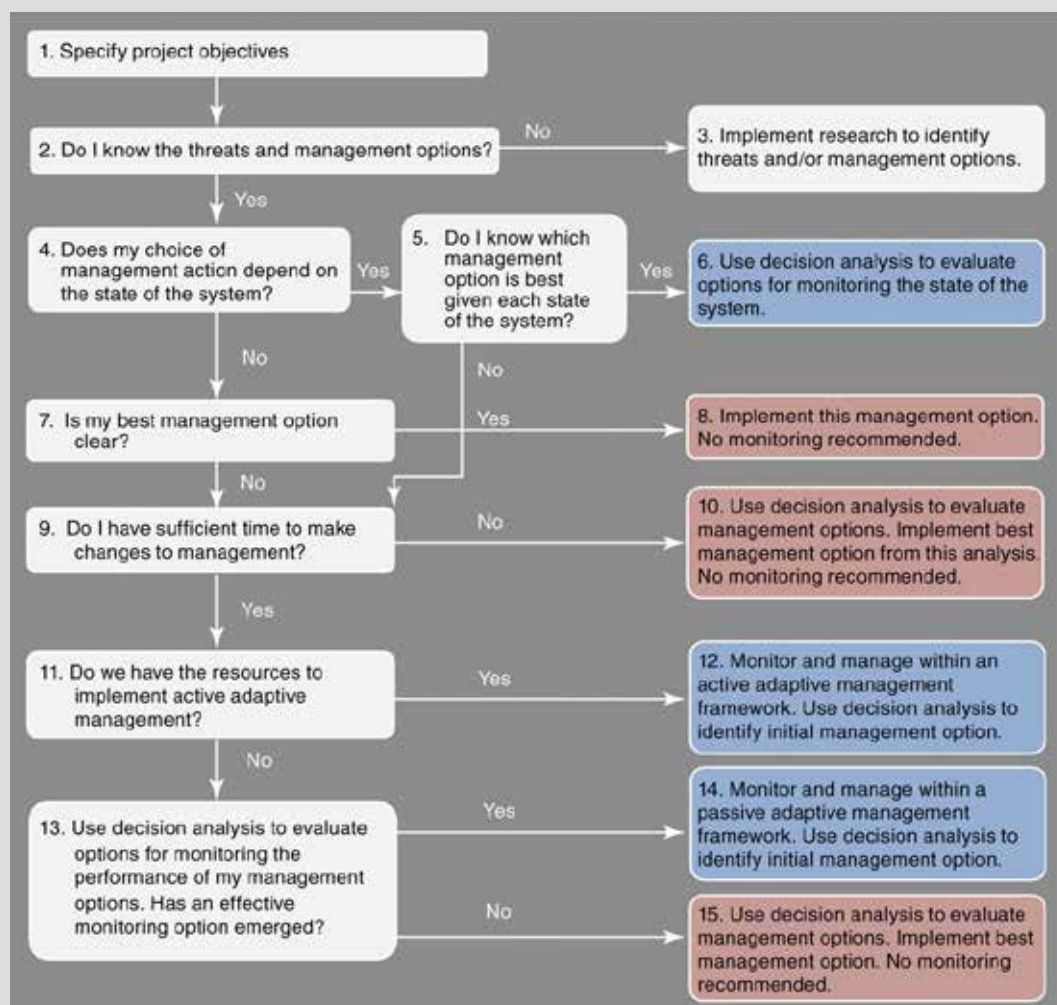
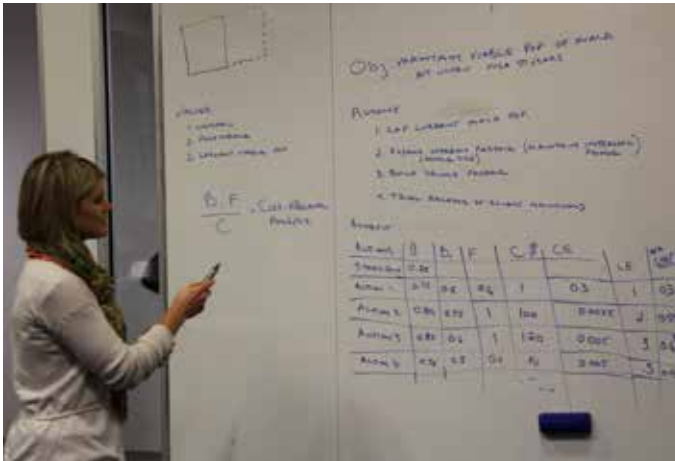


Figure 1: A decision tree for deciding when to implement monitoring to improve conservation management (and when not to). From Macdonald-Madden et al, 2010.



Decision scientist Tara Martin undertakes a 'back-of-the-envelope' cost effectiveness analysis of options for maintaining viable populations of a threatened mammal species in Central Australia (at a Parks Australia Monitoring workshop run at the University of Melbourne in 2012). The analysis basically multiplies the expected benefit of an option by the feasibility it will work divided by the cost (which generates a benefit/cost ratio). When values are informed by experts, in this case Parks Australia managers (pictured below), it can inform effective management strategies in a very short time. (Photos by David Salt)



In the past two decades, a set of tools has been developed to aid governments or other management agencies in spatial planning: identifying where to locate and how to configure areas that should receive some sort of protection to conserve their natural processes. Related tools are used in contexts like urban planning, electoral districting, and school zoning.

4. Recurrent decisions: Many natural resource management decisions are made repeatedly; the same, or a similar, decision is revisited on a recurrent basis. For example, hunting seasons, fishing catch limits, or quotas for subsistence take are often set on an annual basis. Likewise, timber management decisions within a large forest are made recurrently. In some parts of the world, the inundation of wetlands is managed on a seasonal or annual basis to provide habitat for migrating species, encourage primary productivity, or set back succession.

Recurrent decisions are challenging because the systems are dynamic. An action taken in one cycle, which might generate a short-term benefit, changes the trajectory of the system, affecting the actions that can be taken subsequently as well as the benefits that will arise from them. To calculate the long-term benefit of taking an action today, we have to anticipate all the subsequent actions that will be taken over the timeframe of management.

5. Multiple objective trade-off decisions: One of the emblematic environmental issues of the late 20th Century in the United States was the development of the Northwest Forest Plan, which exposed conflicts between conservation of the northern spotted owl and maintenance of an old-growth timber industry. These challenges are ubiquitous in environmental decisions—humans have multiple objectives they would like to achieve from an ecological system, like biodiversity conservation, economic growth, recreation, provision of ecosystem services, and maintenance of subsistence harvest, but it might not be possible to achieve them all to their highest degree. How should these trade-offs be managed?

Multi-criteria decision analysis provides a set of tools for structuring, analyzing, and negotiating multiple-objective decisions. Coupled with the growing field of non-market valuation, these tools can help decision makers grapple with the core values question at the heart of these decisions—what is the relative value of the multiple objectives?

3. Sometimes we need more information, sometimes we don't

Uncertainty surrounds all the decisions we make. It's something we must capture in our predictions and consider in our final decision. Sometimes, reducing that uncertainty can improve our ability to make a decision. However, sometimes additional learning won't change our choice of action. Recent advances provide tools for understanding when new science is needed.

Monitoring: Monitoring is generally viewed as a 'smart' activity in the pursuit of improved conservation outcomes. By explicitly asking the question, "is spending money on monitoring justified?", we must be prepared to not monitor in some cases. Good monitoring rests fundamentally on a clear justification for acquiring information in the first place. That is, what we strive to know should be driven by what we need to know (see [Decision Point #52](#), p4-6, and see the box on analysis and monitoring).

Value of information: Most information gathered about the natural world does not help inform decision-making. This is the crux of value-of-information analysis (see [Decision Point #67](#)). It is about asking whether money and time spent collecting data might change our decision. It recognizes that gathering information costs money, delays actions, and takes resources away from management. Formal value-of-information analysis calculates how much a decision maker's expected outcomes will improve if uncertainty can be resolved prior to committing to a course of action.

Adaptive management: Adaptive management, sometimes called 'learning whilst doing', is a formal approach to making linked decisions in the face of uncertainty. More than just trial and error, adaptive management embeds the idea of value of information into monitoring design and experimental actions, so that management actions can improve over time (see [Decision Point #102](#)).

Management thresholds & seaweed

In order to adaptively manage protected areas, conservation managers need to know when to implement management actions to prevent ecosystems trending towards an unfavourable condition. Whilst ecological research and monitoring can help define unfavourable ecosystem conditions; the question of when to implement a management action requires value judgements by decision-makers. Such judgements require decision-makers to subjectively trade-off competing objectives (eg, there is a trade-off between environmental (eg, biodiversity benefits), social (eg, visitor satisfaction) and economic (eg, the cost of management actions) objectives).

Decision scientists worked with Parks Victoria to trial a structured decision making (SDM) process to explore where to set management thresholds for the intertidal brown alga, *Hormosira banksii*, at Port Phillip Heads Marine National Park (Addison et al, 2015). *Hormosira* (also known as Neptune's necklace) is an indicator of the condition of invertebrate and algal communities on Victoria's rocky intertidal reefs. Parks Victoria has identified that a key threat to intertidal reef communities is trampling by humans.

The challenge for Parks Victoria is this: If the condition of *Hormosira* starts to decline in the future, at what point should a more intensive management strategy be implemented to minimise the impact of trampling?



Researchers worked with Parks Victoria to determine management thresholds relating to the condition of Neptune's necklace, a common seaweed of the intertidal zone. (Photo by Museum Victoria)

The SDM process involved Parks Victoria staff (decision-makers and on-the-ground rangers) and marine scientists with expertise in intertidal ecology. All participants could contribute valuable experience and knowledge of the management of marine national parks and the effectiveness of biodiversity protection.

Participants developed a series of management objectives and alternative management actions relevant to the decision context. The management objectives represented environmental factors (eg, to improve the condition of *Hormosira*), social factors (eg, to improve visitor satisfaction) and economic factors (eg, to minimise resources spent), all of which were considered fundamentally important to the decision context by participants.

Participants were asked to consider the current condition of *Hormosira* and three future scenarios of reduced condition of *Hormosira* (Figure 1). Under each of these scenarios, they elicited participants' estimates of the consequences of management alternatives on the management objectives. The SDM process is particularly useful at this stage, as the researchers can incorporate participants' uncertainty in scientific knowledge, particularly when it comes to predicting the effectiveness of management alternatives under future scenarios.

Reference

Addison PFE, K de Bie & L Rumpff (2015). Setting conservation management thresholds using a novel participatory modelling approach. *Conservation Biology*. DOI: 10.1111/cobi.12544. And see the story on this paper in [Decision Point #74](#)

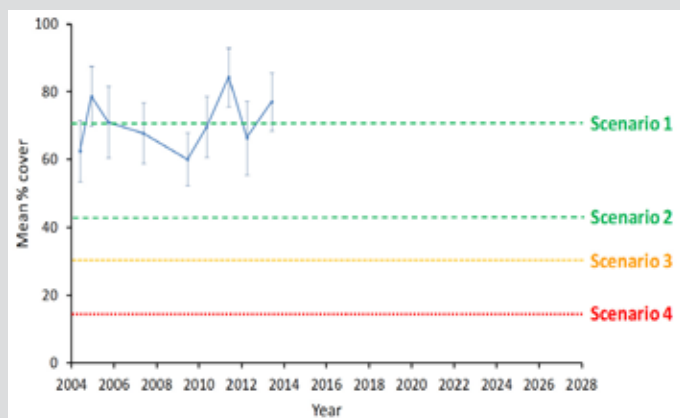


Figure 1: The current condition of *Hormosira* (blue line: mean % cover \pm standard error) at Point Lonsdale intertidal reef (Port Phillip Heads Marine National Park) from 2004 – 2013. Through the SDM process, participants were asked to consider the current condition of *Hormosira* (Scenario 1) and three future scenarios of reduced condition of *Hormosira* (Scenario 2-4), when estimating the consequences of management alternatives on management objectives.

4. Implementation: requires careful attention to the decision, the people, and the institutions involved

Environmental decisions are hard, not only because their elements are complex and multi-faceted, but because the political and institutional settings in which they are made are filled with passionate, contentious and diverse people. Faced with this complexity, decision-analysis tools are very attractive. They give us the hope of structured, rational deliberation. But these tools don't actually make the decisions. They are only aids for the decision makers, not prescriptions. Further, we are as distrustful of other people's processes as we are of their policies, so the proposal to even undertake decision analysis is itself often contentious.

If these reservations can be overcome, however, decision analysis does offer a rich set of tools to enhance the deliberation behind environmental decisions. Overcoming those reservations requires attention to the interpersonal, institutional, and political dynamics surrounding the decision.

Here are three suggestions that may help overcome some of these reservations?

(1) Use rapid prototyping to sketch an initial decision analysis.

Minimal initial investment with the opportunity for substantive input may invite decision makers and stakeholders into the process.

(2) Pay attention to who the decision maker is. In complex institutions (like government agencies), it may not be immediately clear who will end up making the decision.

(3) Recognise the importance of stakeholders and the possibility of multiple decision makers. Deeply political settings involve multiple decision makers, who may not agree to cooperate. Recognition of this dynamic can lead to strategies for negotiation, compromise, and forward progress.

Making good decisions about and for the environment is enormously challenging. Rather than throwing our hands up and saying it's all too hard, decision makers stand to make considerable gains if they can better engage with the rich and growing field of decision analysis.

More info: Eve McDonald-Madden e.mcdonaldmadden@uq.edu.au



Eve McDonald-Madden working with stakeholders at a Parks Australia workshop on monitoring. The first steps in any effective decision making process is to set a context and agree on objectives. This usually involves a blackboard (or whiteboard) or any medium the allows the people involved to share their thinking in an explicit form. (Photo by David Salt)

An annotated bibliography

A guide for anyone wanting to delve further into the field of decision analysis

Risk analysis in environmental decisions

Burgman MA (2005). *Risks and decisions for conservation and environmental management*. Cambridge University Press, Cambridge, UK.

This is the definitive text for risk analysis in environmental decisions. Mark Burgman describes all the tools a decision maker needs to make decisions in the face of uncertainty, including methods for characterizing, eliciting, estimating, and describing uncertainty; methods for evaluating risk tolerance; and methods for managing risk.

Rapid prototyping for environmental decisions

Garrard GE, L Rumpff, MC Runge & SJ Converse (2017). Rapid prototyping for decision structuring: an efficient approach to conservation decision analysis. Pages 46-64 in Bunnefeld N, Nicholson E, Milner-Gulland EJ, eds. *Decision-Making in Conservation and Natural Resource Management: Models for Interdisciplinary Approaches*. Cambridge University Press.

Just as a car manufacturer designs a new model through prototypes, so, too, a decision maker can analyze a decision through a series of successive prototypes. The first prototype is simple, but captures the rough outline of the decision; subsequent prototypes add complexity as needed. Georgia Garrard and colleagues describe the process of rapid prototyping for environmental decisions.

Structured decision making

Gregory R, L Failing, M Harstone, G Long, T McDaniels & D Ohlson (2012). *Structured decision making: a practical guide for environmental management choices*. Wiley-Blackwell, West Sussex, UK.

Robin Gregory and colleagues have been facilitating difficult environmental decision processes for decades. In this text, they synthesize their insights, presenting an outline of decision analysis from an environmental management perspective. Their emphasis is on the foundational tools to achieve a proper framing of the decision problem.

The genesis of decision analysis

Howard RA (1988). Decision analysis: practice and promise. *Management Science* 34:679-695.

In the early 1960s, Ron Howard and Howard Raiffa established the field of decision analysis, pulling together tools and insights from many other fields of study. In this paper, Ron Howard describes the genesis and tenets of the field and reflects on the accomplishments in its first quarter century.

The project prioritization protocol

Joseph LN, RF Maloney & HP Possingham (2009). Optimal allocation of resources among threatened species: a project prioritization protocol. *Conservation Biology* 23: 328-338.

Economists and mathematicians have studied the nature of allocation decisions for decades. The underlying structure of such decisions involves understanding the collective benefits from allocating resources to a portfolio of actions, and developing algorithms to find the optimal portfolio for investment. Liana Joseph and colleagues bring these insights into the realm of environmental decisions. The project prioritization protocol (PPP) is an easy-to-use algorithm that provides an approximately optimal solution for allocating resources.

How values are at the center of all decisions

Keeney RL (1992). *Value-focused thinking: A path to creative decision making*. Harvard University Press, Cambridge, Massachusetts, USA.

In this seminal text, Ralph Keeney describes how values are at the center of all decisions, and how embracing this fact can lead to better decisions. The recognition that there are often many fundamental objectives at play in decisions leads to the need for multi-criteria decision analysis. Although Keeney describes a broader set of decisions than just environmental decision, the tools and insights are particularly apt for the types of decisions environmental managers face.

Adaptive management

Keith DA, TG Martin, E McDonald-Madden & C Walters (2011). Uncertainty and adaptive management for biodiversity conservation. *Biological Conservation* 144:1175-1178.

In 1986, Carl Walters wrote a seminal text on the adaptive management of natural resources. Since then, adaptive management has become a guiding principle, mantra, and buzzword throughout environmental management institutions. In this paper, David Keith and colleagues review the principles of adaptive management and reflect on the lessons learned in the quarter century since its first description.

A framework for the role of monitoring in environmental decisions

McDonald-Madden E, PW Baxter, RA Fuller, TG Martin, ET Game, J Montambault & HP Possingham (2010). Monitoring does not always count. *Trends in Ecology & Evolution* 25: 547-550.

In the 1990s and 2000s, the need for monitoring to accompany environmental management became a dogmatic prescription, but this prescription was not always founded in a full understanding of how the monitoring would affect the desired management outcomes. In this review, Eve McDonald-Madden and colleagues provide a framework for the role of monitoring in environmental decisions, along with a useful decision tree for evaluating when to implement monitoring.

Expert elicitation and value of information in adaptive management

Runge MC, SJ Converse & JE Lyons (2011). Which uncertainty? Using expert elicitation and expected value of information to design an adaptive program. *Biological Conservation* 144:1214-1223.

The value of information is a decision analytic tool to evaluate how much the expected outcomes of a decision could improve if uncertainty could be resolved first. In this paper, Michael Runge and colleagues describe the calculation of value of information, and how it can be coupled with a process of formal expert elicitation. They then show how it was used to evaluate management and research options for an iconic environmental decision.

Decision theory and game theory

von Neumann J & O Morgenstern (1944). *Theory of Games and Economic Behavior*. Princeton University Press. Princeton, New Jersey, USA.

During World War II, an extraordinary amount of thought and research was dedicated to how competing actors make decisions. Out of this work was born both the fields of decision theory and game theory. In this seminal and still vibrant text, John von Neumann and Oskar Morgenstern describe the mathematical foundation of utility theory (to characterize the risk tolerance of a decision maker) and game theory (to characterize decisions made by competing decision makers). For a more recent (and delightfully engaging) introduction to game theory, see Len Fisher's *Rock, Paper, Scissors: Game Theory in Everyday Life* (2008), Basic Books, NY.



The [Decision Point archive](#) has many stories on most of the elements discussed in this special feature. [Issue #74](#) presents several case studies on how structured decision making has been applied to solve real world conservation challenges.



Building models for better decisions

The role of ecological understanding, models & model makers in decision making

By Peter Vesk and Libby Rumpff (University of Melbourne)

Models are basic to good decision making. System models are representations of the dynamics of an ecological system, a conceptual map of how the system works. They enable us to specify our thinking on how the system responds to management. Without them in our decision frame it's unlikely our choices will be well founded. What's more, and just as important, without a system model the potential to learn is limited.

Of course, a good decision can be made without a formal model. However, it should be recognized that however a decision is made there is always at least a 'mental model' guiding the decision maker(s). A mental model is simply someone's thinking about how something works in the real world. Sometimes it's referred to as intuitive decision making; in other words, decisions are based on someone's intuition.

Relying on a mental model can be problematic in that it is not documented, so the underpinning logic can't be interrogated or transferred, and its compatibility with observed data cannot be tested. In addition, there is a greater risk that the model is myopic, and does not consider the suite of objectives or actions relevant to the context of the decision.

Formal models are documented for all to see. They can be qualitative or quantitative, and both have their uses. A qualitative model helps to clarify thinking, and test logic, coherence and communicability. A qualitative model may provide a starting point to sketch out the dynamics of the system as a way to brainstorm and check that the relevant values, threats and drivers, and associated interventions are captured in the model. Sometimes it becomes evident that knowledge is insufficient. The qualitative model provides the initial basis for understanding of structural uncertainty and key knowledge needs. Qualitative models can be subsequently

Above: Restoring native woodlands (right) on degraded land (left) can take considerable resources (time and money). Working without a good explicit conceptual model of how the ecosystem functions or responds to management interventions is a bit like shooting in the dark. Most of the time you miss your target and it's very hard to learn from success or failure.

developed into quantitative models with the accumulation of (expert or empirical) knowledge and data.

Where a qualitative model falls down is in the capacity to be tested or updated with data; how does one accept, reject or modify a model in the light of observations when no quantitative prediction is made?

A model needs balance

Models often have limited uptake out in the real world because of a lack of technical expertise within an agency, cynicism about the value of models in decision making, and an inappropriate balance between complexity and simplicity of a model. Finding a good balance between simplicity and accuracy in models of ecosystem behaviour is a major challenge. A model which represents a particular system well can result in poor generality, and a model that is too simple can result in a poor representation of fundamental ecosystem dynamics. Care should be taken to restrict the number of variables and states to those which are thought to be most important and relevant to the objective at hand. Uncertainty about system processes and responses to management can make this a difficult step, but building and testing multiple models, in addition to sensitivity analysis, can help obtain the right level of model simplicity.

Key messages:

1. System models can provide explicit representations of how an ecosystem functions. In the context of making decisions, this allows a manager to explore the impacts of management actions on the key values (objectives) at hand.
2. Models are sometimes developed in response to a demand for assistance from managers. Models are also developed by researchers to supply a solution to an environmental problem. Supply- and demand-driven models have different development pathways and these differences help explain some of the successes and frustrations in modelling for environmental problems.
3. Clarity about the context of the problem is critical; elements include values (objectives), performance measures, spatial and temporal scales, and alternative actions.
 - a. In the case of a demand for a system model, it is crucial that decision makers and modellers define these elements of the problem context because they will shape the development and specification of the model.
 - b. From a supply side, judging whether a given model is fit for purpose requires evaluating whether a model aligns with each of these elements of the problem context.
4. All models are 'wrong' but some can be useful if they help us improve management and assist us in learning. Understanding the purpose of the model is crucial; at times qualitative models can help creative thinking about how the system works and exploration of key uncertainties; at other times models are required to make quantitative predictions about critical performance measures under alternative management actions and scenarios. Understanding the context is important to managing frustration and providing effective decision support.
5. Models for decision support need to account for uncertainty to allow a decision maker to express their attitude to risk.
6. To be useful in decision support, models need to be understood and accepted by decision makers. This requires good communication about assumptions of structure and function of the models. Development of new models may benefit considerably from decision makers participating in model development.

Models for Structured Decision Making

Increasingly, policy makers and managers are encouraged to adopt structured approaches to addressing environmental problems. An explicit understanding of how ecological systems function (via system or process models) is essential for this for many reasons. Science and system models help to articulate the responses of the ecological objectives of interest, they assist in predicting the possible outcomes of management, and they aid in understanding the uncertainty and knowledge gaps that impede management decisions.

In Structured Decision Making (SDM), modelling has the role of making the predictions that link alternative actions to the desired outcomes (see [Decision Point #104](#)), and to identify whether there is critical uncertainty about the system that prevents making obvious decisions. In filling these needs, system models provide the backbone of any analysis of trade-offs and provides a plan for learning through the process of adaptive management (see [Decision Point #102](#)).

Armed with clarity on the decision maker's objectives and the alternatives available, there should be no barrier to developing a fit-for-purpose model (see [Decision Point #74](#)). However, often this doesn't happen. Why is that? In our experience it has a lot to do with the social dynamics involved in building 'decision support tools' to support the decision-making process. In what follows we will examine this and discuss some of the triggers underpinning the development of system models.

The demand and supply of system models

In order to gain some understanding of the process of modelling to support decision-making, it's useful to examine

Models enable structured learning

Despite significant investment and decades of effort to reverse widespread declines in species and habitat in regional Australia, most indicators are suggesting the situation is only growing worse. What's more, resource management agencies have found it difficult to demonstrate how the investment of public funds has contributed to positive ecological change.

Of course, demonstrating the return on NRM investment is always a major challenge. Management interventions are made over a range of scales in time and space, and the behaviour of the ecological systems we're dealing with is highly complex and variable. The sin is not that we haven't tried to manage for positive change; rather it's that we haven't effectively learnt from our efforts. Ecologists and practitioners have generally failed to undertake structured learning from our successes and failures. Indeed, much data and priceless experience has essentially disappeared. With growing ecological pressures and increasingly limited resources, surely that's unacceptable?

So, what do we need to do to effectively learn? What's required is a management framework with clear objectives, a capacity to assess progress toward those objectives, and the ability to evaluate the extent to which particular management actions contribute to that progress. Adaptive management provides just such a framework and at its centre is a process model of how we think the system works.

some of the triggers for their development. One obvious trigger is when decision makers ask for a model to be developed to help them choose between available options. For this discussion we call this scenario demand-driven.

Supply-driven modelling for environmental problems also exists. Commonly, models are available to describe ecological systems arising from years of ecological research. And when calls to make decisions for an environmental problem arise, the nearest, or most elaborate, or best-fitted models are proposed as suitable for the problem at hand.

Alternatively, researchers may become aware of an environmental problem and identify a knowledge gap or technological opportunity for extending model specification. Arising from discussions around policy for environmental problems, a scientist may respond to what they see as failings of existing knowledge or models concerning the problem at hand by developing models to incorporate processes that they identify as being omitted or ignored.

The funding and rewards for academic researchers tend to emphasize the supply-driven mode of knowledge acquisition. Problems can arise with this when applied to environmental decision making. Equally, the development of models for the SDM process in the demand-driven situation brings its own challenges to be resolved for a model to be fit for purpose (see Table 1).

Continued on page 16

A checklist for an effective model?

How can you be certain you've done a good job of developing a model? Basically, if it helps in better understanding how an ecosystem functions and assists in learning then you're on the right track. Here's a short checklist of what needs to be considered for an effective model. Of course, there's a lot more that goes into an effective model but if you can't check off these five basic points then the long term value of your effort is far from assured.

1. Identify the fundamental objective(s) of the decision maker. What do they really care about having more or less of? Which of these are the focus of your model?
2. Identify some attribute(s) valued by the decision maker that would serve as a direct and sensitive performance variable(s).
3. Ensure your model makes predictions about your performance variable with uncertainty, and include management actions that would enable you to compare the outcome of scenarios and actions on a common, measurable scale. Incorporating uncertainty allows for attitudes to risk to be accommodated.
4. Ensure the spatial and temporal scales are relevant to those of the decision maker, both in terms of specification of performance measures, and the timeframes over which performance is examined; eg, if a manager is focused on national status of a threatened species, the performance measure may examine the potential national distribution of the species, over years to decades to centuries.
5. Ideally, develop your model with the relevant experts and stakeholders (ie, a participatory approach). Have your model understood and accepted by the relevant decision maker and have it reviewed by a peer. Modelling is difficult and it is easy to make mistakes or to have missed inconsistencies in one's model.

Table 1: Issues relating to supply and demand approaches to modelling

Type	Scenario	Potential issues for decision support
Supply	Existing model: A modeller believes a model is fit for purpose for a particular context	<ul style="list-style-type: none"> • The model focuses on different performance measures, or the incorrect spatial/temporal scale • Plausible scenarios and actions may not be represented • Model is perceived as a 'black box' by managers and other end-users • Uncertainty may not be explicitly presented to allow evaluation of which sources of uncertainty are critical
Supply	Non-existing model: A modeller believes they could develop a process or decision model to assist with a technical gap relevant to a problem, or suite of problems	<ul style="list-style-type: none"> • The problem is not clearly specified in terms of one or all of the following steps of structured decision making: <ul style="list-style-type: none"> • Decision context • Objectives and performance measures • Relevant or available alternative management actions • The role of the model is unspecified as a decision support (trade-offs, support monitoring inputs) or process model • Representation of uncertainty in model outputs is unclear or unspecified • Tolerance to uncertainty in model outputs not clear or justified • Model is perceived as a 'black box' if managers not involved in decisions about structure and assumptions
Demand	Non-existing model: A manager/ decision maker requests development of a process or decision model to assist with a technical or knowledge gap relevant to a problem, or suite of problems	<ul style="list-style-type: none"> • Clarity on the role of the model may be unclear: where does it fit in the structured decision making process? Is it instrumental for qualitative or quantitative articulation and specification of the system elements, behaviour and key uncertainties? • Is it a decision model to understand trade-offs? • Is a decision actually on the table? Are the agency participants privy to that decision?

The process in making adaptive management meaningful

Using process models to guide investment in the management of native vegetation

What about models for vegetation change at the scale of the landscape? NRM agencies often have objectives of increasing the extent of native vegetation at a landscape scale. One Catchment Management Authority in Victoria had an objective to increase the extent and condition of native vegetation across the catchment (a region of around 5,000km²). What is an appropriate system model for this? (Rumpff et al, 2010).

Vegetation condition may be assessed with multiple measures of abundance and diversity of structural components of woodland vegetation (eg, numbers of large trees, shrub cover, native species understorey, and the like) though not many people would naturally think in terms of those multiple attributes at once. However, state and transition models* (STMs) have been an important tool for managers to represent the possible changes that might occur within a landscape, and think about possible transitions between the states in which it can exist (Figure 1 shows a process diagram of how the STM is developed. Figure 2 shows the resulting conceptual state-and-transition model for non-riparian woodlands). We reasoned that this STM framework was useful for communicating (or reporting) the state of the system. But underlying this we utilized the multiple attributes within a Bayesian network.

A great strength of this approach was that the models could be initially specified from the scientific literature and expert information. The models could then be updated with observational data. Bayesian updating is a great match to an adaptive management program: the initial parameterization of the model represents existing hypotheses about how the system works, with uncertainty represented and probability distributions. As data from experiments and monitoring are

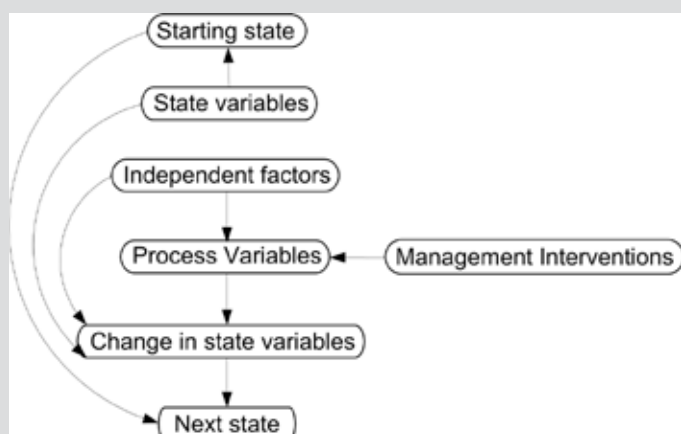


Figure 1: A process diagram for iterating states and transitions (from Rumpff et al, 2010a). The states are discrete states of vegetation structure and composition that can be identified in the landscape, which are defined by various attributes of vegetation condition (ie, state variables). Independent factors control the conditions at the site, but cannot be modified at the site scale. Process variables are controlling environmental variables that can be modified at the site-scale by management actions.

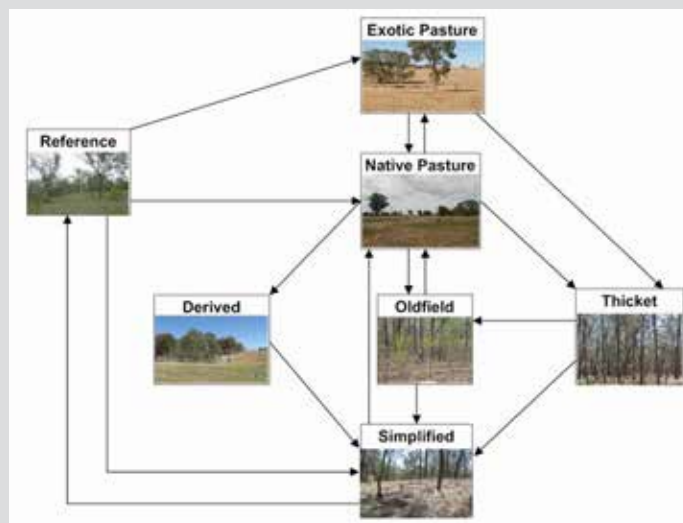


Figure 2: A conceptual state-and-transition model for non-riparian woodlands (adapted from Rumpff et al 2010a). State changes are determined by a change in the value of individual vegetation attributes. (Note, management interventions, threats and state variables are not shown on this figure. The states are described in Rumpff et al 2010.)

accumulated, the probability distributions are changed and thereby the model changes. This can allow evaluations of outcomes under different cases or situations or actions.

The model was non-spatial, it included multiple attributes that comprised elements of a complex multiple objective. It included management actions that might be employed by landholders and managers.

After just one monitoring/learning cycle, seven years after the first investments, we found that updated models predict markedly different transition probabilities compared with initial models based on expert opinion. This has strong implications for the cost-efficiency of restoration strategies. The STM approach provides a sound theoretical basis for restoration decisions, while the Bayesian network implementation provides a workable framework for using the STM adaptively.

**A modelling approach which is becoming increasingly common in native vegetation management are State-and-transition models (STMs). These models describe different states of vegetation condition that may occur, and the possible transitions that may occur between these states. They are popular because there is recognition that there are different states of vegetation condition in the landscape with different land-use histories, multiple non-linear pathways of change between states, threshold behaviour, and possible barriers to restoration. Their adoption signalled a move away from the traditional views of Clementsian succession, whereby an initial management intervention (such as fencing or tree planting) will theoretically result in a single linear successional trajectory towards a 'climax' or reference vegetation community.*

Continued from page 14

Complex environmental challenges usually take considerable time to resolve. It's not uncommon for these challenges to result in the development of many models answering different needs as different groups and decision makers become involved. Some of the models developed will likely be demand-driven, meeting the requirements of different groups and different decision makers. Some models will be supply-driven, developed by researchers attempting to understand the nature of the challenge. Such was the case when it was proposed that management should thin degraded woodlands in Victoria to restore some of their natural values. We were involved in the development of some of the models that were used to assist with decision making surrounding this challenge.

The following case study on thinning for conservation demonstrates many of the issues we have discussed here regarding demand- and supply-driven models, and how different managers may have different decision scope and values at hand.

Navigating the development of a fit-for-purpose model: Modelling thinning-for-conservation outcomes

Box-ironbark forests and woodlands in Northern Victoria have been degraded through a history of timber exploitation, gold mining and pastoralism. Many of these forests have a structure characterized by dense stands of small, even-aged stems, few habitat resources for dependent fauna and sparse, low-diversity plant understories. How do you restore some of the natural values to these degraded forests?

Drawing on a history of thinning from production forestry, it was suggested that cutting down some trees in crowded stands might help restore a more natural forest structure. We became involved in thinning for conservation when Parks Victoria was implementing an experimental thinning trial. Parks Victoria wanted to model the consequences of thinning actions on public land for conservation objectives. Our part in this journey



A thinned eucalypt stand showing the recovery of thick understory vegetation some 6 years after thinning. (Photo by Chris Jones)

Hunting down hawkweed

A model for alpine success

Effectively detecting an invasive species during its earliest phase of introduction can save a lot of future damage and expense. Yet this is also when the species is most difficult to detect, lurking at a low density in a large landscape. So, how much should we be willing to spend searching for these 'needles in haystacks'? Decision researchers Cindy Hauser and Michael McCarthy developed a pest surveillance model that shows how we can best design searches that keep costs down (Hauser and McCarthy 2009).

Some of the factors their model needed to deal with include:

Probability of occurrence: Some locations will be more likely to harbour the pest than others. Habitat suitability models can map where the pest is most likely to find a happy home, and can incorporate dispersal to predict where the pest is most likely to be at a particular time.

Ease of detection: Some habitats are easier to search than others. We need to understand how a habitat's terrain and the search effort employed affect our chances of successfully detecting the pest.

Benefits of detection: What will be gained if we find and treat the pest now rather than later? Economic, biodiversity and other values may indicate that some areas are more important for protection from the pest than others.



A volunteer seeks hawkweed in the 2012 detection experiment. (Photo by Roger Cousens)

started as a small program of research that has continued now for more than a decade.

At the heart of the matter was the question: would thinning these forests result in improvements in the population status of native plants and animals (compared to continuing timber harvest or doing nothing)? Later, we worked with another agency of the Victorian government on a similar question, but arising from private landholders seeking funding through land stewardship schemes (Bush Tender) to conduct thinning activities on private land, notionally for conservation outcomes.

We developed and used several models over this time, including tree diameter growth models, non-spatial and static logic trees

of wildlife population viability, expert-parameterized stand dynamics simulators, statistical models of stand density and vegetation condition, and decision trees. Each of these models had slightly different performance variables, at different scales, with different treatment of space and time.

- The logic trees of population viability concerned single animal species and the probability that a $R > 1$ (positive population growth rate) could be supported on such a stand, given the species habitat requirements for foraging and breeding. These models were demand-driven; they were developed in response to a request from Parks Victoria. [Terry Walshe developed these models

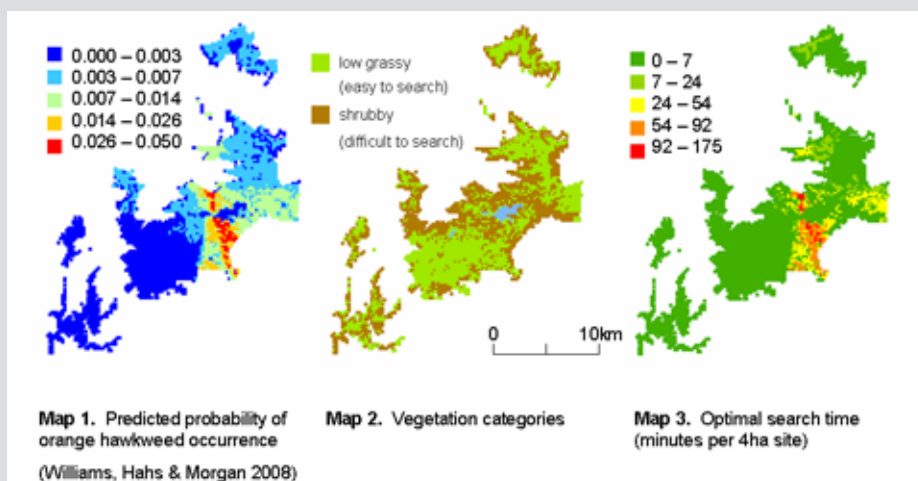


Figure 1: Modelling effective control of an invasive weed is more than just knowing where it occurs. It's also about cost-effectively detecting it. (From Hauser & McCarthy 2009)

They tested their model on how it might help eradicate orange hawkweed (*Hieracium aurantiacum*) an invasive daisy which poses a major threat to Australian grasslands and temperate areas.

Researchers have previously predicted the occurrence of orange hawkweed on the Bogong High Plains in Victoria using measures of vegetation, wetness, disturbance and wind dispersal (Map 1). These predictions are vital for prioritising surveillance efforts.

To use the model for surveillance planning approach, we also need information about detection. Researcher Nicholas Williams split the landscape into 'easy to search' and 'difficult to search' vegetation categories and estimated detection rates based on his searching experience (Map 2; Williams, Hahs and Morgan 2008).

Hauser and McCarthy then conducted 'detection experiments' with hidden hawkweeds and volunteer searchers, allowing for more rigorous modelling. Their method indicates that effort should be concentrated in a small proportion of sites with a high probability of orange hawkweed occurrence, though it also depends on the vegetation type (Map 3).

It is worthwhile visiting grassy sites, at least briefly, even when the probability of presence is quite small because they are easy to search. However, difficult-to-search shrubby sites tend to be surveyed either thoroughly or not at all. Their inclusion in the surveillance plan requires a high probability of pest presence and large detection benefits.

with faunal experts. The log trees utilised combinations of and/or statements to define the necessary and substitutable resources for population growth. These were coupled with expert-specified probabilistic statements of the supply of those habitat attributes in different forest states. The result being probabilistic statements about the likelihood of viable populations of the yellow footed antechinus, diamond firetail, brown treecreeper and sugar glider.]

- The stand dynamics models made predictions through time of proportions of simulated stands with desirable stand structures, principally high vs low density of large stems. These models were supply-driven, being developed to address shortcomings of the former, static models. [These models were developed by Chrissy Czembor and utilized state-and-transition models. Czembor worked with individual experts to define the state and transition agents that could affect the change of the state of the forest from one state to another, at each timestep. These transition agents included: growth, coppice growth, fire, windthrow and management actions including thinning operations. Uncertainty in the models was rigorously accounted via Monte-Carlo sampling routines in the software, eliciting ranges for parameters, and through different experts specifying different parameter values.]
- The statistical models (see the box on 'to thin or not to thin') characterized relationships between vegetation condition or quality—the abundance and richness of native and exotic understorey plants—and stand density, and with effects from thinning actions. These were again demand-driven, arising in response to a request for information about private land thinning activities.

Those various models made different predictions about the likely benefits of thinning actions. The logic trees of population viability and the stand dynamics models both revealed such uncertainty, that no clearly identifiable winning action could be identified. By careful accounting of uncertainty, these illustrated that the decision needed to account for a decision maker's attitude to risk. If a decision maker was risk-averse, worried about getting fewer large trees 100 years hence, they would be best advised to not conduct any cutting. But if they were aspirational, favouring the best outcome with little regard to how likely it was, a manager would favour ecological thinning.

The statistical models of vegetation condition indicated that thinning could improve some aspects of the native plant diversity, but could also benefit weedy grasses—not a good outcome. They also specified the dependence of the outcome of the land use of the site and implications for seedbanks and soil condition.

How did those models perform or what purpose did they serve? The models served to highlight aspects of the problem: performance variables (stand structure, viable wildlife populations, aspects of vegetation condition) relevant to values; causal structure of the forest ecosystem and management interventions. The stand dynamics models highlighted the long lags in forest responses to intervention and large contribution of disagreement among experts about the fundamental processes driving stand dynamics in these forest and uncertainty about the rates at which some of processes occurred (eg growth rates, probability of cut stems resprouting). By accounting uncertainty and highlighting aspects most critical to resolve, these aided maximizing the chance of

To thin or not to thin

Understanding dense eucalypt stands and the pros and cons of thinning

In Victoria, there is an increasing call for management of dense eucalypt stands on both private and public land. The most commonly cited management option is thinning – cutting down a proportion of stems and applying herbicide to prevent regrowth. The theory is that the release from competition should make the remaining stems grow faster, larger, and broader, as well allowing the recovery of understorey vegetation.

The questions for managers then are: How bad is a dense stand for biodiversity and what is the benefit of thinning? But perhaps more importantly, we need to ask whether thickets pose a problem that warrants major investment from government. At what scale would thickets need to be a dominant form to cause concern for those species, and communities, and at what scale is the treatment cost effective?

Chris Jones and colleagues sought answers to these questions using data from two separate field projects conducted in box-ironbark woodlands and forests in central Victoria, where they evaluated the vegetation structure of dense regrowth stands of eucalypts, and the effect of thinning management (Jones et al, 2015). In order to determine what density of stems and cover of understorey 'should' be expected in natural systems, they evaluated their results in relation to published benchmarks of stem density and understorey vegetation cover.

They found that stands with stem density greater than benchmark levels suppress native understorey vegetation cover below its benchmark levels. Thinning stems can restore native understorey vegetation (richness and cover) in the short term, providing the soil seedbank has not been removed and there is no excessive grazing. This is the desired outcome from thinning, but the catch is that BOTH native and exotic species can recover following thinning.



Dense regrowth of grey box (*Eucalyptus microcarpa*) on a grazing property in central Victoria. (Photo by Chris Jones)

Is my model fit for purpose?

Matching data and species distribution models to applications

Species distribution models (SDMs) are becoming a fundamental tool in environmental decision making. For instance, SDMs are used to identify areas suitable for reintroduction of threatened species, sites at risk of biological invasions or to direct the search for new populations of species.

There are many considerations involved in building useful correlative SDMs. For an SDM to have good predictive ability we need to identify critical environmental predictors. For example, do average temperature, average rainfall and soil pH accurately capture why this plant species happens here and not there?

Defining a suitable extent for the model is also fundamental. Am I interested in describing the habitat preferences for this mammal species at a continental scale, or do I want to understand its preferences at a local scale? There is a lot written about these and other aspects of building SDMs (and Brendan Wintle has developed an excellent checklist of the basics in Decision Point #67 [insert link http://decision-point.com.au/article/developing-and-interpreting-species-distribution-models/](http://decision-point.com.au/article/developing-and-interpreting-species-distribution-models/)).



Species distribution models are used for all sorts of purposes in conservation planning and management. For example, they have been used to understand the invasion of cane toads in Australia. (Cane toad image by Ben Phillips).

An important consideration is that the type of data available for a species affects the interpretation and reliability of SDM outputs? This is a critical aspect in the practice of species distribution modelling yet it's often overlooked. Users often underestimate the strong links between data type, model output and suitability for end-use. Species distribution models can lead to suboptimal conservation outcomes and misguided theory if the underlying data are not suited to the intended application.

desirable outcomes for wildlife and plants (see [Decision Point #105](#)).

The statistical models were able to be incorporated into a decision tree to guide how the agency may choose to respond to landholder bids for funds from conservation incentive schemes. It enabled a reflection of the decision maker's longer-term desire to learn about the outcome of interventions so as to respond to funding bids more effectively in the future.

If you'd like to read more about the challenge of thinning for conservation, see Czembor et al, 2009 and Jones et al, 2015 (and see the box on 'to thin or not to thin').

On reflection, were the models effective in resolving the problem? It is possible to draw the conclusion that the models are not being used by the responsible agencies to justify a widespread decision on thinning activities, and so they have failed in being useful to the resolution of the environmental problem of high stem density in degraded forests. That is a harsh and rigid evaluation. As laid out early in this article, models play different roles in different contexts.

We were engaged to conduct research and modelling by agency staff at some remove from actual decision makers (ministers and elected representatives or deputy secretary of government agencies). A decision was probably not on the table, rather the research was commissioned to clarify the problem, prompting consideration of the scope of the problems, scales for consideration, necessary values and performance variables.

Did we address each of the potential issues in decision-support tools (Table 1)? No, but by framing and solving 'toy' versions or facsimiles of the problem of high stem density and the potential role of thinning, the decision problem becomes better understood. In hindsight there were situations where perhaps more effort to work with our partners to identify the

real decision to be made, and the real decision maker, might have helped us address the right objectives, measures, scales etc. Perhaps that is because we were learning as we went, and trying to be play multiple roles: decision analysts and scientists and modellers.

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References

- Czembor CA & PA Vesk (2009). Incorporating between-expert uncertainty into state-and-transition simulation models for forest restoration. *Forest Ecology and Management* 259: 165-175.
- Jones CS, DH Duncan, L Rumpff, FM Thomas, WK Morris & PA Vesk (2015). Empirically validating a dense woody regrowth 'problem' and thinning 'solution' for understory vegetation. *Forest Ecology and Management* 340: 153-162.
This paper was discussed in [Decision Point #86](#)
- Hauser CE & MA McCarthy (2009). Streamlining 'search and destroy': cost-effective surveillance for pest management. *Ecology Letters* 12: 683-692. You can read a discussion on this work in [Decision Point #31](#), p4-5.
- Williams NSG, AK Hahs & JW Morgan (2008). A dispersal-constrained habitat suitability model for predicting invasion of alpine vegetation. *Ecological Applications* 18: 347-359.
- Rumpff L, DH Duncan, PA Vesk, DA Keith & B Wintle (2010). State-and-transition modelling for Adaptive Management of native woodlands. *Biological Conservation* 144: 1224-1236.
This story was discussed in [Decision Point #47](#), pages 4-6.
- Guillera-Arroita G, JJ Lahoz-Monfort, J Elith, A Gordon, H Kujala, PE Lentini, MA McCarthy, R Tingley & BA Wintle (2015). Is my species distribution model fit for purpose? Matching data and models to applications. *Global Ecology and Biogeography* 24: 276-292. This paper was discussed in [Decision Point #97](#)



Prioritising environmental investments

Why bother? How is it done?

By David Pannell (University of Western Australia)

In most environmental programs around the world, the funding provided by governments falls well short of that needed to deal comprehensively with the environmental issues in question. Success rates of 5-20% are common in competitive funding rounds for environmental projects.

As a result, program managers cannot avoid the need to decide which potential investments should receive funds and which should not. Even where no explicit prioritisation process is used, managers are implicitly prioritising, although probably not systematically or transparently.

Key messages:

Prioritisation is unavoidable; most environmental programs have too few resources to meet their goals.

Given the great variation between potential environmental investments, good prioritisation is critically important.

In most cases, good prioritisation of environmental investments is reasonably easy to apply.

In some cases, prioritisation is more complex and difficult, requiring special techniques.

Despite the compelling case for good prioritisation, it is often not practiced.

Above: A group of CEED ecologists and economists with an interest in landscape restoration are shown a variety of restoration projects in the Canberra region. Many restoration proposals are put forward for funding but available resources will only support a small percentage of these. How do decision makers prioritise which projects should get funding and which don't? (Photo by David Salt)

The benefit that can be generated by systematic prioritisation depends in part on how heterogeneous the various investment options are. The greater the diversity in costs and benefits amongst different projects, the more important it is to accurately identify the best projects. The variance in these factors is often extremely high. It's not uncommon for Benefit: Cost Ratios of different proposed projects to vary enormously. For example, the data set used by Fuller and colleagues (2010) for 7000 potential environmental investments reveals Benefit: Cost Ratios that vary by more than eight orders of magnitude.

Reinforcing that finding, we estimated the environmental gains that are possible through high quality prioritisation of investments relative to poor-quality prioritisation (and compared this with random project selection). High quality prioritisation can give you a gain of 50 to 100% relative to poor-quality prioritisation and a gain of up to 800% relative to random project selection (Pannell & Gibson, 2016).

Of course, there are additional costs involved in undertaking good-quality prioritisation processes, relative to simpler approaches. However, the estimated benefits are easily large enough to justify the additional effort.

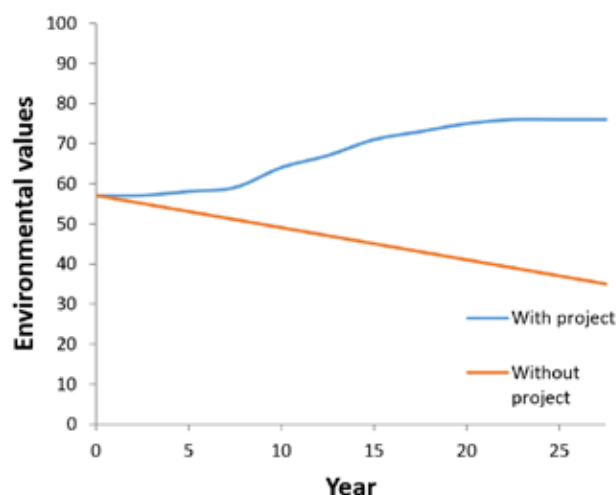


Figure 1: Comparing what happens 'with versus without' the project.

So if you are going to engage in a prioritisation process, what do you do? In the following pages I outline the basic approach. If you would like more detail on any aspect, have a look at Pannell (2015) as it sets out a relatively simple and plain-speaking discussion on the process (though in a lot more detail).

Generating the greatest environmental benefit

The starting assumption is that the objective of the prioritisation process is to generate the greatest environmental benefits for the community as a whole by allocating a limited budget across a range of potential projects. In other words, the organisation wishes to maximise the value for money from its environmental investments.

Sometimes environmental organisations seek to rank locations, or issues, or desired outcomes, with no explicit project activities defined. This is problematic because value for money depends on the answers to questions like, "what is the technical feasibility of generating the hoped-for benefits?"; "to what extent would the community cooperate?" and "what would it cost?" Those questions can only be answered for a particular set of actions or interventions – a project. Prior to ranking projects,



What is the benefit and cost of an environmental restoration project? What are the important assumptions you need to make in estimating the costs and benefits? These are important questions to answer when prioritising which projects will be funded and which won't. (Image courtesy of Greening Australia.)

The wrong metric for ranking projects

Around the world there are thousands of different quantitative systems to rank projects. At the heart of most of these systems is a formula or metric that combines various pieces of information about a project to produce a number that provides an overall assessment. There are various errors that can be made when putting together a ranking metric, and the quality of the results is quite sensitive to some of the common errors. These errors include: weighting variables inappropriately; adding variables that should be multiplied; comparing outcomes without considering counterfactuals (ie, ignoring what might happen if the project hadn't happened, the outcome might have occurred anyway); omitting key variables related to benefits; ignoring costs; and measuring activity (outputs) instead of outcomes. In this article we outline the basic approach that will ensure you avoid these errors.

each potential project needs to be clearly defined in terms of what would be done, where, and by whom.

Let's begin with the simplest case, where each project is independent of other projects – the benefits and costs of a project do not depend on which other projects are implemented. We will deal with more complex cases later.

One crucial insight is that to estimate the benefits of a project you need to know the benefit values 'with the project' and the values 'without the project' (both of which usually have to be predicted).

Comparing values 'with versus without' is not the same as comparing values 'before versus after' the project. The reason is that conditions may not be static in the absence of the project. For example, it may be that an environmental asset would degrade in the absence of the project (as illustrated in Figure 1). Remarkably, a study looking at 17 existing systems from around the world for prioritising conservation projects found that only one correctly used the with-versus-without approach to estimate benefits (Maron et al, 2013).

A simple benefit-cost metric

Here is a Benefit-Cost Ratio metric for ranking environmental projects. It is the simplest theoretically defensible formula that should be used: *Equation 1*

Equation 1

$$BCR = \frac{[V(P_1) - V(P_0)] \times A \times (1 - R)}{C + M}$$

where: **BCR** is the Benefit: Cost Ratio. The higher the *BCR*, the better the project,

V() represents the values (or benefits or services) generated,

P₁ represents the outcomes with the project in place,

P₀ represents the outcomes without the project in place,

A is the level of adoption/compliance (by individuals or businesses whose cooperation is needed to achieve the project's goal) as a proportion of the level needed to achieve the project's goal,

R is the probability of project failure – in other words, the riskiness of the project,

C is the total project cash costs, and

M is total discounted maintenance costs.

$[V(P_1) - V(P_0)]$ in the above formula represents the difference in overall values with versus without the project (assuming full compliance, $A=1$, and zero project risk, $R=0$). It is the potential benefit of the project if everything goes right.

V can be measured in monetary terms, or in some other unit that makes sense for the types of projects being ranked.

The structure of this formula is very important. Benefits (in the numerator) are divided by costs. The three main parts of the top row are multiplied together, not added, because the overall benefit is proportional to each of these parts. There are no weights applied to any of these variables. And costs (in the denominator) get added up, rather than multiplied.

One framework for robust prioritisation: INFFER

There are thousands of different systems in use to rank environmental projects for funding. Unfortunately, judging from the many examples I have examined, most of the systems in use are very poor. Indeed, the performance of many of them is not much better than choosing projects at random. If only people would be more logical and thorough in their approach to ranking environmental projects! The potential to reduce wastage and improve environmental outcomes is enormous. Attempting to get managers, researchers, policy people and decision makers to appreciate this has been a major driving force behind much of my work in environmental economics. It led to the creation of the INFFER framework (Pannell et al, 2013) and it also has sparked many editorials on my blog, [Pannell Discussions](#).



INFFER team members assess proposed environmental projects with stakeholders in North Central Victoria. (Photo by Geoff Park)

This simple formula can be modified in a number of ways to better deal with some of the complexities decision makers will face in the 'real' world. Here are two such modifications. The first incorporates factors dealing with time lags and discount rates. The second provides a more nuanced engagement with risk.

Incorporating time: Equation 2

Equation 2

$$BCR = \frac{[V(P_1) - V(P_0)] \times A \times (1 - R) / (1 + r)^L}{C + K + M}$$

where: **L** is the lag time in years until most benefits of the project are generated,

r is the annual discount rate, to account for the fact that money spent on the project incurs the equivalent of an interest cost, and **K** is the total project in-kind costs of the organisation that is running the project, not costs to people whose behaviour the project is intended to influence.

The last part of the numerator, " $/(1 + r)^L$ ", is included to discount future benefits back to their present value. It is important to include this part of the formula if different projects vary substantially in the time lags until they generate benefits. The choice of discount rate, r , can make a large difference to the estimated benefits for projects with very long-term effects.

Incorporating risk: Equation 3

Equation 3

$$BCR = \frac{[V(P_1) - V(P_0)] \times A \times (1 - R_t) \times (1 - R_s) \times (1 - R_f) \times (1 - R_m) / (1 + r)^L}{C + K + (E + M) \times (1 - R_f)}$$

where: **R_t**, **R_s**, **R_f** and **R_m** are the probabilities of the project failing due to technical risk, socio-political risks, financial risks and management risks, respectively, and

E is total discounted compliance costs. These are involuntarily borne private costs, where people are forced to comply by regulation or similar. We recommend that private costs that are borne voluntarily should not be included, because the fact that there is voluntary cooperation indicates that the costs are offset by unmeasured private benefits.

Simplifications

There are a number of simplifications in the above formulae, even for the most complex of them:

- Assuming that benefits are linearly related to the proportion of people who adopt the desired new practices or behaviours;
- Representing project risks as binary variables: success or complete failure;
- Having only one time lag for all benefits from the project;
- Approximating the private benefits and voluntary private costs as zero; and
- Treating the project costs, maintenance costs and compliance costs as if there was only one combined constraint on their availability.

Simplifications are essential to make the system workable, but care is needed when selecting which simplifications to use. Each of these simplifications can be relaxed if desired.

Dealing with uncertainty

Uncertainty and knowledge gaps are unavoidable realities when evaluating and ranking projects. The available information is almost always inadequate for confident decision making. Key information gaps often include: the cause-and-effect relationship between management actions and environmental outcomes; the likely behavioural responses of people to the project; and the values resulting from the project.

Although uncertainty is often high, the ranking procedure used remains important. Even given uncertain data, the overall benefits of a program can be improved substantially by a better decision process. Indeed, benefits appear to be more sensitive to the decision process than to the uncertainty. For example, we found that there is almost no benefit in reducing data uncertainty if the improved data are used in a poor decision process (Pannell & Gibson, 2016). On the other hand, even if data is uncertain, there are worthwhile benefits to be had from improving the decision process.

This is certainly not to say that uncertainty should be ignored. Once the decision process is fixed up, uncertainty can make an important difference to the delivery of benefits.

There are economic techniques to give negative weight to uncertainty when ranking projects. However, we suggest a simpler and more intuitive approach: rating the level of uncertainty for each project; and considering those ratings subjectively when ranking projects (along with information about the Benefit: Cost Ratio, and other relevant considerations).

Apart from its effect on project rankings, another aspect of uncertainty is the question of what, if anything, the organisation should do to reduce it. It is good for project managers to be explicit about the uncertainty they face, and what they plan to do about it (even if the plan is to do nothing). Simple and practical steps could be to: record significant knowledge gaps; identify the knowledge gaps that matter most through sensitivity analysis (Pannell, 1997); and have an explicit strategy for responding to key knowledge gaps as part of the project, potentially including new research or analysis.

In practice, there is a tendency for decision makers to ignore uncertainty when ranking projects, and to proceed on the basis of 'best-guess' information, even if the best is poor. In support of that approach, it is often argued that we should not allow lack of knowledge to hold up action, because delays may

result in damage that is costly or impossible to reverse. That is reasonable up to a point, but sometimes organisations are too cavalier about proceeding with projects when they have little knowledge of whether they are worthwhile. It may be at the expense of other projects in which they have much more confidence, even though they currently appear to have lower BCRs. It is not just a question of proceeding with a project or not proceeding – it's a question of which project to proceed with, considering the uncertainty, benefits and costs for each project. When you realise this, the argument based on not letting uncertainty stand in the way of action is rather diminished.

In some cases, a sensible strategy is to start with a detailed feasibility study or a pilot study, with the intention of learning information that will help with subsequent decision making about whether a full-scale project is worthwhile, and how a full-scale project can best be designed and implemented. A related idea is active adaptive management, which involves learning from experience in a directed and systematic way (see [Decision Point #102](#)).

Particularly for larger projects, we believe that one of these approaches should be used as they have great potential to increase the benefits that are generated. They imply that the initial ranking process should not produce decisions that are set in stone. Decisions may need to be altered once more information is collected. We should be prepared to abandon projects if it turns out that they are not as good as we initially thought, rather than throwing good money after bad.

In the environment sector, managers are almost never explicit about the uncertainties they face, there usually is no plan for addressing uncertainty, projects are funded despite profound ignorance about crucial aspects of them, proper feasibility assessments are never done, active adaptive management is almost never used, and ineffective projects that have been started are almost never curtailed so that resources can be redirected to better ones. In these respects, the environment sector is dramatically different from the business world, where people seem to be much more concerned about whether their investments will actually achieve the desired outcomes. Perhaps the difference is partly because businesses are spending their own money and stand to be the direct beneficiaries if the investment is successful. Perhaps it is partly about the nature of public policy and politics. Whatever the reason is, there is an enormous missed opportunity here to improve environmental outcomes, even without any increase in funding.

The choice between the three versions of the ranking formula depends on the importance of the issues being addressed, the scale and costs of the projects being considered, the time and resources available for the ranking process, and the availability of the information needed for each formula.

Measuring benefits

In the BCR formulas presented in equations (1), (2) and (3), the numerators represent the benefits of a project. The equations show that the benefits are determined by several factors: the value or importance of the environmental benefits generated (measured as a difference, with versus without the project); the level of adoption or compliance with the project (ie, the extent to which the necessary actions are actually taken); various risks that may cause the project to fail; and the time lag until benefits

arise. If we represent the risks as probabilities of failure, the numerator represents the 'expected' benefits, using 'expected' in the statistical sense of a weighted average.

If the projects being prioritised all produce benefits that are similar in nature, and policy makers are happy to base their measure of benefits on scientific criteria, benefits (ie, V in the above formulae) can be measured using ecological criteria that are specific to the issue. The advantages of using monetary values are that it allows you to:

- (a) compare value for money for projects that address completely different types of issues (eg, river water quality versus recreational benefits versus income) and
- (b) assess whether a project's overall expected benefits exceed its total costs.

Economists have developed a range of methods that can be used to monetise environmental values: so-called non-market valuation methods. While these methods are not without their challenges and problems, they do have some strengths. One is that they allow the preferences of the broader community to be transparently considered during environmental prioritisation. They don't rule out using an approach that combines community preferences with those of experts. The methods have been subjected to deep scrutiny and testing, and clear guidance on preferred procedures for implementing them and analysing the results are available. They result in a more logically consistent and defensible set of weightings than are often used in weighting processes that avoid monetisation.

Notwithstanding the advantages of monetising the benefits, it can be challenging to obtain appropriate values. Help from an expert is often advisable.

Measuring costs

Measuring the costs of environmental projects is conceptually simpler than the measurement of benefits, but it is not without its challenges. Many environmental prioritisation processes fail to include the full range of relevant costs, particularly maintenance costs (Armsworth 2014). If benefits are to be counted for a long time frame (eg, decades), then any relevant maintenance costs should be counted over the same time frame. Similarly to the benefits, maintenance costs should be discounted to present values, to avoid over-stating their significance. (In principle, even the initial 3-to-5-year costs of establishing a project should be discounted as well, but failing to discount over such short time frames is a less serious error than failing to discount maintenance costs.)

Let us now consider a few more more complex prioritisation problems.

Multiple versions of the same project

There are always many different ways of designing a project, and they can vary greatly in value for money. Therefore, it can be worth evaluating more than one project per asset or issue, especially in important cases. For example, we may have identified that it is a high priority to invest in protection of a particular environmental asset (a wetland, or a particular species, or a river), but there remains the issue of how ambitious the project should be. If there is currently a 20% probability that a species will go extinct over the next 20 years, should the project aim to reduce that probability to 10%, 5%, 1%, 0.01%, or what? Should a project that addresses water pollution from agricultural nutrients aim to reduce nutrient inflows to the water body by 10%, 20%, 40% or 80%? Project options such as these can be compared by defining a separate project for each target level, and comparing the BCR for each option.

Doing this comparison can be important because the BCRs can vary widely depending on the target chosen. A key factor behind this is the empirical observation that project costs are often related to the target in a highly non-linear way, with costs escalating greatly at higher targets. For example, Figure 2 shows the estimated costs of reducing phosphorus pollution in the Gippsland Lakes in eastern Victoria, depending on the percentage reduction. Clearly, the cost increases at an increasing rate as the target reduction is increased.

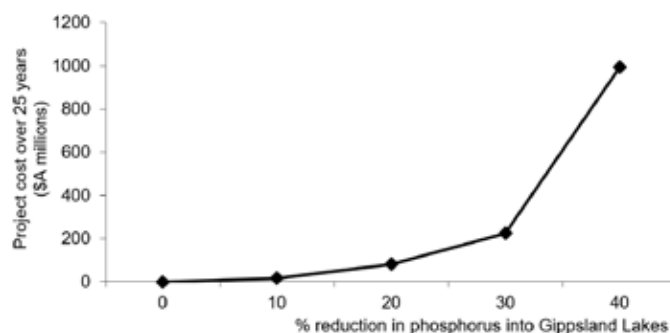


Figure 2: The estimated costs of reducing phosphorus pollution in the Gippsland Lakes (Victoria), see Roberts et al, 2012.

When comparing distinct projects, the way to generate the most valuable environmental benefits for the available resources is to select those projects with the highest BCRs, up to the point where the budget is exhausted. However, when comparing multiple versions of the same project, the criterion is a little different. It is to select the most ambitious project that has a BCR above the threshold level for acceptance. The threshold depends on how tight funding is, and on the performance of other competing projects.

Multiple benefits from one project

The prioritisation formulas (1), (2) and (3) discussed earlier are designed to work where there is a single type of benefit from a project, or where the values for multiple benefits have already been converted into a common currency, such as dollars, and added up. If a project has multiple benefits and they are not monetised, the other option is to combine them by weighting them (to reflect their relative importance) and adding them up (see Pannell 2015). This is essentially what monetising them does, but sometimes people have a prejudice against using monetised values in this process.

Prioritisation when projects depend on each other

The formulas are also founded on an assumption that the projects being compared do not depend on each other. This means that they cannot be used, for example, to compare which parts of a region should be restored through revegetation, because the benefits of such revegetation depend on what vegetation there already is, and on whether other parts of the region are revegetated. Sound optimisation in this situation requires a more complex approach, such as a mathematical model that optimises across the whole region.

A concluding comment: People often respond to the manifest inadequacy of budget allocations to the environment by demanding we should spend more. Yet, consider this. If you could double your budget for projects by putting a bit more effort into your project ranking process, would you do so? Of course you would. Doubling the environmental benefits generated from your environmental investments is rather like doubling your budget (but much much easier to achieve). If your current ranking system is of the usual questionable quality, doubling the benefits (or more) is readily achievable using the approaches advocated here.

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Reference

Armstrong PR (2014). Inclusion of costs in conservation planning depends on limited datasets and hopeful assumptions. *Annals of the New York Academy of Sciences*. doi: 10.1111/nyas.12455
Read a discussion on this paper in [Decision Point #87](#)

Fuller RA, E McDonald-Madden, KA Wilson, J Carwardine, HS Grantham, JEM Watson, CJ Klein, DC Green & HP Possingham (2010). Replacing underperforming protected areas achieves better conservation outcomes. *Nature* 466: 365-367.
Read a discussion on this paper in [Decision Point #41](#), p2-5

Joseph LN, RF Maloney & HP Possingham (2009). Optimal allocation of resources among threatened species: a project prioritisation protocol. *Conservation Biology* 23: 328-338.
Read a discussion on this paper in [Decision Point #29](#), p8-10

Maron M, JR Rhodes & P Gibbons (2013). Calculating the benefit of conservation actions. *Conservation Letters* 6: 359-367.
Read a discussion on this paper in [Decision Point #69](#)

Pannell DJ (1997). Sensitivity analysis of normative economic models: Theoretical framework and practical strategies. *Agricultural Economics* 16: 139-152.

Pannell D, AM Roberts, G Park & J Alexander (2013). Designing a practical and rigorous framework for comprehensive evaluation and prioritisation of environmental projects. *Wildlife Research* 40: 126-133. <http://dx.doi.org/10.1071/WR12072>

Pannell DJ (2015). *Ranking environmental projects*, Working Paper 1506, School of Agricultural and Resource Economics, University of Western Australia, Crawley, Australia. <http://ageconsearch.umn.edu/handle/204305>
Read a discussion on these papers in [Decision Point #75](#), p4,5

Pannell DJ & FL Gibson (2016). The environmental cost of using poor decision metrics to prioritise environmental projects. *Conservation Biology* 30: 382-391.
Read a discussion on this paper in [Decision Point #82](#)

Roberts AM, DJ Pannell, G Doole & O Vigiak O (2012). Agricultural land management strategies to reduce phosphorus loads in the Gippsland Lakes, Australia. *Agricultural Systems* 106: 11-22.

Institutional challenges to good prioritisation

Our experiences in encouraging environmental agencies and other relevant bodies to use sound and rigorous approaches to prioritisation have shown that a number of institutional challenges can arise.

One issue is that it isn't necessarily apparent to people that their organisation is using a prioritisation process with serious weaknesses, nor that there is an opportunity to deliver substantially greater environmental outcomes by improving the process. It is common for organisations to develop a prioritisation process without an understanding of the essential principles outlined here. Sometimes considerable effort is put into developing the process, and this results in a strong commitment to the process and a belief that it is sound, despite serious flaws. Convincing organisations that there would be substantial benefits in modifying the process can be difficult in this situation, despite evidence of the magnitude of potential gains.

Contributing to a reluctance to change in some cases is concern about the greater cost of a more rigorous prioritisation process. Where the existing process is superficial and highly subjective, advice to include more and stronger evidence may not be welcomed. And yet, the improved gain from using a strong prioritisation process versus a weak process (Pannell & Gibson, 2016) seem easily large enough to justify slightly higher costs from a more comprehensive approach – yet some are still difficult to convince.

Apart from the financial cost of doing a more thorough prioritisation process, it can also be more demanding in terms of time and data. In some organisations there is a culture of rushing decision making, such that there is insufficient time allowed for data collection and analysis to support the decisions. This often goes hand in hand with a view (or at least a rationalisation) that subjective judgements are sufficient. However, subjective judgements in ignorance of the principles outlined here are highly unlikely to deliver the hoped-for environmental outcomes.

In some cases, decision makers prefer to use a process that is not as comprehensive and transparent as we have recommended, because it reduces their flexibility and their scope to select 'pet projects' that are actually not good investments.

Some people in the environment sector shy away from using approaches that they judge to be too tainted with economics, because they believe that economic drivers have caused environmental problems, so economic thinking should be avoided when trying to solve them. This unfortunate prejudice, when applied to prioritisation, simply results in poorer environmental outcomes.

Perhaps related to this is a reluctance in some cases to prioritise at all, apparently based on a feeling that to do so is defeatist. This often arises when discussions of conservation triage arise which is really just a form of prioritisation (Joseph et al, 2009). Opponents to the process say we should never 'give up on a species', we should save all species. While this might be a laudable goal, it ignores what is actually happening in the real world in which constrained government (and NGO) budgets simply can't save all species. If we were to engage in robust prioritisation we could save more species than we currently do.



There are many institutional challenges to good prioritisation. One common problem is that once an organisation has adopted a process for ranking projects it is often loathe to change it even when it can be shown a stronger prioritisation process will deliver significant gains. (Photo by David Salt)



Valuing information for better environmental decisions

Does more data necessarily lead to better outcomes?

By Jonathan R Rhodes (University of Queensland)

“We need more data before we can make a decision!” Everyone working around environmental management would have heard someone say this at some stage. Indeed, most scientists faced with an environmental challenge will say something similar. And yet, if you think about it, the logic of this statement is flawed because doing nothing and collecting data is, in itself, a decision!

The alternative to this decision (to do nothing and collect data) is to do something (take a management decision) with the limited information currently available. Yet these alternatives are rarely evaluated and compared for the outcomes they produce: would collecting new data change and improve the management decision compared to not collecting any new data?

Key messages:

1. Collecting more data is often used as an excuse to delay a decision.
2. Sometimes new data is important to a decision, sometimes it won't affect it.
3. Value of information (Vol) analysis can assist decision makers assess the value of collecting more information.
4. Vol analysis was originally developed in the decision sciences as a means of explicitly calculating the improvement in decision outcomes when new information is first collected prior to making a decision.
5. Vol Analysis provides an important toolbox for improving adaptive management processes.

Above: Many species of migratory bird are in trouble with populations in steep decline. There are loud calls for governments to act to save them, but there is so much we don't know about the likely responses to management actions. The result is often a call for more information, more data. Is the collection of new data likely to make a difference? Value of information analysis is one approach that seeks to find an answer to this question. (Photo by Rob Clemens)

These are exactly the types of decision problems that a technique called Value of Information (Vol) analysis can help us with. In this article, I'll look at this method and how it can be applied to environmental decisions. I'll provide some of the theory behind the approach, work through a simple case study problem to illustrate how the analysis works, and also look at a couple of examples where Vol analysis has been employed. The aim is to make Vol analysis more accessible to scientists and managers involved in environmental decision-making.

Data, uncertainty and decision-making

Statisticians and ecologists have spent a lot of time thinking about and measuring how much additional data is needed to reduce uncertainty. Data on species' distributions and trends, environmental threats, and the impact of climate change all provide information that reduces uncertainty about environmental systems. But, from a decision making point of view, this information is only valuable if it results in better decisions being made. More information will almost always reduce uncertainty, but it may not necessarily provide the right information to improve decisions, or may not be relevant information for improving decisions. In these cases it may be better to invest resources in management actions right now, rather than in collecting more data (which also uses up resources).

Although it may sound counter-intuitive, there are two main reasons why collecting more data may not improve a decision, even though it reduces uncertainty. First, the best decision may be so obvious that, even with a lot of uncertainty, we are almost certain which action is the best one to choose anyway even with existing uncertainties. For example, it may be so much more cost effective to recover a koala population through reducing mortality than to restore habitat, that reducing uncertainty about the effects of each these actions on population persistence may have little effect on the decision (Maxwell et al, 2015). If this is the case we may make the same decision regardless of whether we collect new data or not and so the management benefit of this data is low.

The other main reason is that the expected benefit of alternative actions may be so similar that it really doesn't matter which action you choose. In this case, collecting more data is unlikely to improve the outcomes of our decisions because, even if it causes us to switch from one action to another, the outcomes will be almost identical (Pannell & Glenn, 2000).

Somewhere in between these extremes, new data and information will be most valuable. However, this intuition doesn't help us identify whether new data is valuable for a specific problem, or which type of data we should collect. This is where Vol analysis comes to the rescue.

Does better information save more koalas?

What's the financial benefit of resolving management uncertainty

Vol analysis was used to evaluate the benefits of resolving uncertainty surrounding the declining koala population in the Koala Coast region of south-east Queensland (Maxwell et al, 2015). We modelled the effectiveness of koala management using current levels of information and compared this to a situation where all uncertainty about birth and death rates, and the effect of forest cover on these rates, was resolved.

We found the optimal management strategies with and without new information on birth and death rates, and the effect of forest cover on these rates, to be very similar. This similarity suggests that resolving uncertainty will have negligible effects on management performance.

Indeed, we found that a 0.034% improvement in the population growth rate is the best we could expect if uncertainty was resolved. When we converted values of information, in terms of population growth rate, into values of information in terms of dollars, we found that if resolving uncertainty costs more than 1.7% of the koala management budget, it would be more cost-effective to allocate that money to direct management action now.

The value of information was low because optimal management decisions were not sensitive to the uncertainties they considered. Decisions were instead driven by a substantial difference in the cost efficiency of management actions. The value of information was up to forty times higher when the cost efficiencies of different koala management actions were similar.

The researchers demonstrated that the value of reducing uncertainty is highest when it is not clear which management action is the most cost efficient.

What is Value of Information analysis?

Vol analysis was originally developed in the decision sciences (Raiffa & Schlaifer, 1961) as a means of explicitly calculating the improvement in decision outcomes when new information is first collected prior to making a decision. It does this by calculating outcomes assuming that an optimal decision is made without collecting new data and comparing this to outcomes assuming an optimal decision is made after collecting new data.

Vol analysis has been applied extensively in the medical/health and economics literature and practice. It has resulted in significant refinements in these fields. Examples in medicine include evaluating medical diagnostic technologies and optimising clinical trials. It has been demonstrated to improve economic values per patient.

In the environmental sciences there has been a history of using Vol to assess information to inform pollution and toxicology control in a health context (Yokota & Thompson, 2004), but use in the conservation sciences (and more broadly for environmental management) has been a more recent development (Colyvan, 2016).



When it comes to managing koalas in the Koala Coast region of Queensland, a Vol analysis showed little benefit was derived from spending extra time and money on gathering information on birth and death rates, and the effect of forest cover on these rates. (Photo by Liana Joseph)

The disciplines of environmental science and management have a long history of thinking about adaptive management (Walters, 1986). This framework aims to improve decision making over time as new information is collected (see [Decision Point #102](#)). In this context, Vol analysis provides an important toolbox for improving adaptive management processes by allowing us to actually value the new information we collect and to choose between different monitoring activities to inform management during the adaptive management process.

Formulating a Value of Information Analysis

As we stated above, Vol is the difference between optimal decisions made with new information and optimal decisions made without that information. Here we are going to write down the formulation for a general Vol problem. There is a bit of maths in it, but for many environmental or ecological problems the associated calculations can be done in a spreadsheet (Canessa et al, 2015).

First of all, think about a case where we are making decisions under uncertainty and we want to know how much better we would do if we eliminated uncertainty entirely (ie, if we were to make decisions under 'perfect' information). But, I hear you say, you will never be able to eliminate uncertainty entirely in a real problem. This is true, but thinking of things in this way does provide useful information on the upper bounds for the benefits of information gain and is often the starting point for a Vol analysis. The improvement in outcomes when eliminating uncertainty entirely is what is known as the Expected Value of Perfect Information (EVPI) and calculating that is what we are going to focus on first.

Expected Value of Perfect Information (EVPI)

Based on the concepts discussed above, the very first thing we need to do is calculate the value of making an optimal decision without new information. But before doing that, let's try to make this a bit more concrete by thinking of an example where we want to choose between two management actions, fire management and habitat restoration to conserve an endangered bird species. We think that fire management will affect breeding success and habitat restoration will affect adult mortality, but we are unsure about how much and so the best management action to choose to maximise population growth rates is also uncertain.

In a simple model we could represent this by having two uncertain parameters representing: (1) the amount that fire management increases breeding success, and (2) the amount that habitat restoration reduces adult mortality. In a more general sense we will often have a set of uncertain parameters, s , that link the management actions to an outcome. In the presence of these uncertain parameters we can calculate the value of making an optimal decision as follows

$$\max_a E_s [V(a, s)] \quad (1)$$

where $V(a, s)$ is the environmental outcome (eg, population growth rate) under action a and parameter values s (eg, the parameters for the effects of fire management and habitat restoration on breeding success and adult mortality), E_s indicates taking the expectation of $V(a, s)$ over all possible values of the uncertain parameter values s (essentially the mean

value), and \max_a indicates finding the action that maximises this expectation (mean). This formulation says that we first find the expected (mean) outcome across all uncertainties and then find the management action that maximises environmental outcome.

Now let's use the same reasoning to calculate the value of making an optimal decision when we have no uncertainty (ie, we have perfect information). In this case, the value can be calculated as follows

$$E_s [\max_a V(a, s)] \quad (2)$$

Here you can see that we first find the management action that maximises $V(a, s)$ for each specific parameter values for s (ie, assuming we know it is the true value) before taking the expectation (mean). Note that we still need to take the expectation after finding the optimal management action because a priori we still don't know what the true parameter values are.

Since EVPI is the difference in value with and without perfect information then we can calculate EVPI as the difference between equation (1) and equation (2), such that

$$EVPI = E_s [\max_a V(a, s)] - \max_a E_s [V(a, s)] \quad (3)$$

Expected Value of Partial Perfect Information (EVPPI)

We can actually extend this idea of the value of perfect information to the idea of eliminating uncertainty in only some parameters and not others. This leads to the idea of the expected value of partial perfect information (EVPPI), which is sometimes referred to as the expected value of perfect X information (EVPXI). This might be useful, for instance, if we wanted to know the value of learning about the effect of fire management on breeding success versus learning about the effects of habitat restoration on adult mortality.

Again, more generally, we can think of this as learning about a subset of the parameters in the set of parameters s . Let's assume that we are interested in the value if eliminating uncertainty is a subset of the parameters, Φ , and the rest of the parameters, Ψ , remain uncertain. The equation for EVPPI based on this is

$$EVPPI = E_\Psi [\max_a E_\Phi [V(a, \Phi, \Psi)]] - \max_a E_{\Phi, \Psi} [V(a, \Phi, \Psi)] \quad (4)$$

where the full set of parameters $s = (\Phi, \Psi)$. Here, in the second term on the right hand side of equation (4) we take the expectation across uncertainty in both Φ and Ψ , and then find the optimal management action to calculate the optimal outcome with uncertainty in both Φ and Ψ . In the first term on the right hand side of equation (4) we take the expectation across uncertainty in Φ (the set of parameters we want to value the elimination of uncertainty from) after finding the optimal action to calculate the optimal outcome with uncertainty only in Ψ . Note that the value of EVPPI will always be less than EVPI since we are only partially reducing uncertainty when considering EVPPI.

Which uncertainty?

Using Vol to design adaptive management

Natural resource management is plagued with uncertainty of many kinds, but not all uncertainties are equally important to resolve. The promise of adaptive management is that learning in the short-term will improve management in the long-term; that promise is best kept if the focus of learning is on those uncertainties that most impede achievement of management objectives. In this context, an existing tool of decision analysis, the expected value of perfect information (EVPI), is particularly valuable in identifying the most important uncertainties.

Expert elicitation can be used to develop preliminary predictions of management response under a series of hypotheses, as well as prior weights for those hypotheses, and the EVPI can be used to determine how much management could improve if uncertainty was resolved.

These methods were applied to management of whooping cranes (*Grus americana*), an endangered migratory bird that is being reintroduced in several places in North America (Runge et al, 2011). The Eastern Migratory Population of whooping cranes had exhibited almost no successful reproduction through 2009. Several dozen hypotheses can be advanced to explain this failure, and many of them lead to very different management responses. An expert panel articulated the hypotheses, provided prior weights for them, developed potential management strategies, and made predictions about the response of the population to each strategy under each hypothesis.

Multi-criteria decision analysis identified a preferred strategy in the face of uncertainty, and analysis of the expected value of information identified how informative each strategy could be. These results provide the foundation for design of an adaptive management program.



A whooping crane in flight in Texas. Vol analysis has been important in determining which is the best strategy to follow in managing this endangered migratory bird. (USDA Photo by John Noll, CC 2.0)

Expected Value of Sample Information (EVSI)

Earlier we mentioned that the assumption of a complete elimination of uncertainty was pretty unrealistic. In fact, we usually go out into the field and collect data that helps reduce uncertainty but not eliminate it. For example, we may collect field data on breeding success in areas where fire is being managed and where fire is not being managed to estimate the effects of fire management on breeding success. Although collecting more data of this kind will reduce uncertainty, it will never remove uncertainty entirely.

Fortunately, EVPI and EVPPI can be extended to deal with this issue and more realistically estimate the value of reducing uncertainty through the collection of data. For this we use the idea of the expected value of sample information (EVSI) and although we do not go into detail of the formulation, for those interested, the equation for EVSI is

$$EVSI = E_s \left[\max_a E_{s|x} [V(a, s)] \right] - \max_a E_s [V(a, s)] \quad (5)$$

where x is the data (sample information) collected. Here, the second term of the right and side of equation (5) is the same as for EVPI. On the other hand, the first term on the right hand side calculates the expectation of the outcomes for each management action over all values of s assuming the data are known, and then maximises this expectation.

Finally, it takes the expectation of this value over all possible values for the data, as the data are also uncertain, to provide an estimate of the value of an optimal decision taken after collecting new data.

Applying Vol analysis

To illustrate the idea of EVPI and EVPPI, let us apply it to our hypothetical threatened bird management problem. Let's assume that we want to choose the management action that maximises the population growth rate of the species and we have a very simple model of population growth that depends on adult female survival, juvenile female survival, and breeding success. We are going to assume that this model looks like this (Pulliam 1988)

$$\lambda = s_A + s_J \beta \quad (6)$$

where λ is the growth rate, s_A is the annual probability of adult survival, s_J is the annual probability of juvenile survival, and β is the probability of successfully breeding (ie, breeding success). We assume that adults can give birth to a maximum of one juvenile (ie, clutch size = 1).

Now, since adult survival and breeding success depend upon the management action, the growth rate is a function of the management action adopted so that

$$\lambda(x_{\text{habitat}}, x_{\text{fire}}) = s_A(x_{\text{habitat}}) + s_J \beta(x_{\text{fire}}) \quad (7)$$

where $x_{\text{habitat}} = 0$ if habitat is not restored and $x_{\text{habitat}} = 1$ if habitat is restored, and $x_{\text{fire}} = 0$ if fire is not managed and $x_{\text{fire}} = 1$ if fire is managed. Here you will see that adult survival now depends on whether habitat is restored or not and breeding success

Table 1: The assumed probabilities of different possible outcomes under do nothing, restore habitat, and manage fire actions. The parameter values for adult survival, s_A , juvenile survival, s_J , and breeding success, β , and population growth rates, λ , under each possible outcome, as well as the expected outcomes for λ under each management action.

Management action	Probability of outcome	s_A	s_J	β	λ
Do nothing	1	0.7	0.5	0.5	0.95
				Expected growth rate	0.95
Restore habitat	0.5	0.8	0.5	0.5	1.05
	0.5	0.9	0.5	0.5	1.15
				Expected growth rate	1.10
Manage fire	0.8	0.7	0.5	0.7	1.05
	0.2	0.7	0.5	0.95	1.175
				Expected growth rate	1.075

depends on whether fire is managed or not. In our hypothetical example, we only have resources to restore habitat or manage fire, and not both, so we also need to work with this constraint:

$$x_{\text{habitat}} + x_{\text{fire}} = 1 \quad (8)$$

It may be reasonable to assume that we have a pretty good understanding of the current survival and breeding success parameters if it were a well-studied population. But we may be uncertain about the likely outcomes under the different management actions; maybe this aspect has never been studied in this population?

But, there may be evidence from other populations, or from experts (Runge et al, 2011), that give us some information a priori about what might happen under the different management actions and to be able to assign probabilities to each possible outcome.

To illustrate this idea, let's assume that under the restore-habitat action there is a 50% chance that we get an adult survival probability of 0.8 and a 50% chance that we get an adult survival probability of 0.9. Then, under the manage fire action there is an 80% chance that we get a breeding success of 0.7 and a 20% chance that we get a breeding success of 0.95. These possible outcomes are summarised in Table 1, together with the expected values (ie, the weighted mean of the possible outcomes, weighted by the probability that each outcome occurs) for the do-nothing, restore-habitat, and manage-fire actions. This reveals that the action that maximises the expected growth rate is to restore habitat, with an expected growth rate of 1.1 (this is, the value you obtain by applying equation (1) to this problem and is the value for the second term of the EVPI equation; the expected outcomes with uncertainty).

But note that, although managing fire doesn't provide such good benefits on average as restoring habitat, there is a small probability that you do better than you can ever do by restoring

habitat (ie, a 20% chance you get a growth rate of 1.175). Now we will apply EVPI and EVPPI to this problem.

In Table 1 we show that the value of the second term of the EVPI equation is 1.1. But what about the first term, which is the outcome when uncertainty is resolved (equation (2))? To calculate this we need to assess what the best strategy under each combination of possible outcomes under the habitat restoration and fire management actions would be. This is summarised in Table 2 showing the optimal strategy for each combination of possible outcomes and also the expectation of the optimal growth rates (after an optimal decision has been made) across the combinations is calculated. This expectation is the first term in the EVPI calculation and, with a value of 1.115, the EVPI is $1.115 - 1.1 = 0.015$, or around 1.5%.

That is to say, if uncertainty were resolved entirely, there would be an improvement in management outcomes that would increase the growth rate by a further 1.5% compared to the case where uncertainty is not resolved.

We can extend this analysis to EVPPI and use it to determine whether it would be best to eliminate uncertainty in the effect of habitat restoration on growth rates or to eliminate uncertainty in the effect of managing fire on growth rates.

The second term in the EVPPI equation is the same as for EVPI, so we know that is still 1.1, but the first term changes. Let's first look at the first term when resolving uncertainty in habitat restoration. In this case, we calculate the growth rate for each potential outcome for the effect of habitat restoration and compare it to the expected outcome for managing fire (acknowledging the effects of managing fire remains uncertain).

This allows us to identify the optimal strategy, under each possible outcome, for the effect of habitat restoration, assuming the effect of managing fire remains uncertain, and then expected value of these optimal outcomes is the first term in the EVPPI calculation (Table 3). Table 3 shows that the first term is 1.1125, so EVPPI for resolving uncertainty in the effect of habitat restoration is $1.1125 - 1.1 = 0.0125$ (or 1.25%). A similar

Table 2: For each combination of potential outcomes under the habitat restoration and manage fire actions, the probability of the outcomes, the optimal action and the optimal growth rate. The expected growth rate of the optimal decisions taken for each is also shown.

Outcome	Probability of outcome	λ under do nothing	λ under habitat restoration (assumed known)	λ under manage fire (assumed known)	Optimal action	Optimal λ
$s_A = 0.8$ under habitat restoration & $\beta = 0.7$ under manage fire	$0.5 \times 0.8 = 0.4$	0.95	1.05	1.05	habitat restoration or manage fire	1.05
$s_A = 0.8$ under habitat restoration & $\beta = 0.95$ under manage fire	$0.5 \times 0.2 = 0.1$	0.95	1.05	1.175	manage fire	1.175
$s_A = 0.9$ under habitat restoration & $\beta = 0.7$ under manage fire	$0.5 \times 0.8 = 0.4$	0.95	1.15	1.05	habitat restoration	1.15
$s_A = 0.9$ under habitat restoration & $\beta = 0.95$ under manage fire	$0.5 \times 0.2 = 0.1$	0.95	1.15	1.175	manage fire	1.175
					Expected growth rate	1.115

Table 3: For each combination of potential outcomes for the effect of habitat restoration, the probability of the outcomes, the optimal action, and the optimal growth rate assuming the effects of managing fire is uncertain. The expected growth rate of the optimal decisions taken for each is also shown.

Outcome	Probability of outcome	λ under do nothing	λ under habitat restoration (assumed known)	λ under manage fire (assumed uncertain)	Optimal action	Optimal λ
$s_A = 0.8$ under habitat restoration	0.5	0.95	1.05	1.075	manage fire	1.075
$s_A = 0.9$ under habitat restoration	0.5	0.95	1.15	1.075	restore habitat	1.15
					Expected growth rate	1.1125

Table 4: For each combination of potential outcomes for the effect of managing fire, the probability of the outcomes, the optimal action, and the optimal growth rate assuming the effects of habitat restoration is uncertain. The expected growth rate of the optimal decisions taken for each is also shown.

Outcome	Probability of outcome	λ under do nothing	λ under habitat restoration (assumed uncertain)	λ under manage fire (assumed known)	Optimal action	Optimal λ
$\beta = 0.7$ under manage fire	0.8	0.95	1.1	1.05	restore habitat	1.1
$\beta = 0.95$ under manage fire	0.2	0.95	1.1	1.175	manage fire	1.175
					Expected growth rate	1.115

analysis can be done for resolving uncertainty in the effect of managing fire on growth rates (Table 4) and the EVPI for resolving uncertainty in this case is $1.115 - 1.1 = 0.015$ (or 1.5%)

The interesting thing you will notice with the analysis above is that resolving uncertainty in the effect of fire management on growth rates is the same as resolving all uncertainty (EVPI) and equal to 1.5%. The reason for this is that, once we have resolved uncertainty in the effect of fire management, there is no longer any uncertainty in which management action to take.

Table 2 shows that uncertainty in the decision is eliminated because: (1) if we know that $\beta = 0.7$ under fire management, then the best strategy is always to restore habitat, regardless of the still uncertain effect of habitat restoration on growth rates, and (2) if we know that $\beta = 0.95$ under fire management, then the best strategy is always to manage fire, regardless of the still uncertain effect of habitat restoration on growth rates.

On the other hand, if we resolve uncertainty in the effect of habitat restoration on growth rates the choice of management decision still depends on the still uncertain effects of fire management on growth rates (Table 2) and EVPI is lower at 1.25%.

This demonstrates that when considering the collection of new data for management, we need to consider the effect on the choice of management decision and not just the effect on uncertainty in our understanding of the system.

In the boxes through this story we have briefly highlighted some recent applications of Vol analysis for decision-making

for conservation. The models they use and problems they solve are more complex than the simple illustration above, but the concepts are identical.

The business of valuing information

Decisions relating to conservation are challenging. They often involve high stakes (get it wrong and you might lose a species or ecosystem), inadequate funding and high levels of uncertainty. As I discussed at the beginning, decision makers often request more information to reduce uncertainty before committing to a course of action – before making a decision.

But, as I hope I have made clear in this short article, putting off a decision is actually a decision itself. Delaying a management action to collect more information involves important trade offs: the cost of gathering that information (in time and money) and the potential improvement in the environmental value you are managing for. And if that value is a critically endangered species, then delay or re-routing funds for management to monitoring might see an irreversible loss if the species goes extinct.

Vol analysis enables decision makers to put a value on what might be gained (or lost) by gathering more information. Undertaking a Vol analysis requires a little technical knowhow but it's a relatively tractable and straightforward process.

While Vol analysis is only beginning to be applied to conservation problems, the approach is already widely used in the corporate world. When it comes to decisions surrounding big investments, business leaders will quickly analyze the value

Quality trumps quantity

The value of information for conservation planning under sea level rise

Here's a slight variation on the theme of value of information. This case study, carried out by Rebecca Runting and colleagues, compared different types of information on sea level rise to determine which type gave the best conservation outcomes. They compared how good a conservation plan was depending on whether it was based on expensive high quality (high resolution) data versus using lower quality (low resolution) data. The high quality data costs a lot more to acquire meaning less money is available for purchasing sites for the reserve network. If planners used the cheaper low quality data, they could purchase more sites for the reserve network.

Their analysis (Runting et al, 2013) came up with an amazing result. Their results suggest that for the upper sea-level rise scenario it was worth spending up to 99% of the available budget on acquiring the high quality data. The break-even cost for the mid-range and lower sea-level rise scenarios was also a large proportion of the total budget (with a mean of 82% and 64% respectively).

In other words, whilst adopting a more accurate approach may mean less land is acquired in terms of overall area (because you've spent some of your money on modelling and data), the land that is acquired would be of greater conservation value than the cheaper approaches. This is because the cheaper approaches tend to omit areas of important conservation



Mangroves on Stradbroke Island, Queensland. Accurately modelling how sea-level rise will change these landscapes could be critical to saving them. (Photo by Catherine Lovelock)

value in the prioritisation process. Sometimes, using these less accurate outputs meant that you could never achieve the same conservation value (no matter how much money you spent), as the areas that are important for conservation didn't exist on the map!

Valuing information for management

Improving the future of box-ironbark forests with targeted learning

Value-of-information analysis reveals the expected benefit of reducing uncertainty to a decision maker. Will Morris and colleagues (Morris et al, 2017) performed this analysis on management of box-ironbark forests in Victoria. With three management alternatives (limited harvest/firewood removal, ecological thinning, and no management), managing the system optimally (for 150 years) with the original information would, on average, increase the amount of forest in a desirable state from 19% to 35% (a 16-percentage point increase). Their Vol analysis revealed that resolving all uncertainty would, on average, increase the final percentage to 42% (a 23-percentage point increase). However, only resolving the uncertainty for a single parameter was worth almost two-thirds the value of resolving all uncertainty.



A red ironbark woodland in Jackass Flat Nature Conservation Reserve, Victoria. What's the value of acquiring more information for our management of these ecosystems?
(Photo by Melburnian, CC BY-SA 3.0)

of acquiring new data as opposed to simply going ahead with the investment decision. For example, deciding on whether to proceed with a new mine requires data from geological surveys and exploratory drillings and so forth. Delaying the decision (to proceed or look elsewhere) comes at a very explicit cost, as does the acquisition of the extra data. The importance of Vol analysis in these situations is clear and they are commonly undertaken.

Given this, there is little excuse for environmental managers not to value the alternatives of whether to proceed with existing information, or to delay a decision when, it comes to deciding on actions to conserve our irreplaceable biodiversity. So, maybe next time you hear someone say: "We need more data before we can make a decision," you might ask on what justification such a delay is warranted.

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References

- Canessa S, G Guillera-Arroita, J Lahoz-Monfort, D Southwell, D Armstrong, I Chadès, R Lacy, S Converse & O Gimenez (2015). When do we need more data? A primer on calculating the value of information for applied ecologists. *Methods in Ecology and Evolution* 6:1219-1228.
- Colyvan M (2016). Value of information and monitoring in conservation biology. *Environment Systems and Decisions* 36:302-309.
- Maxwell SL, JR Rhodes, MC Runge, HP Possingham, CF Ng & E McDonald-Madden (2015). How much is new information worth? Evaluating the financial benefit of resolving management uncertainty. *Journal of Applied Ecology* 52:12-20.
Read a discussion on this paper in [Decision Point #87](#)
- Morris WK, MC Runge & PA Vesk (2017). The value of information for woodland management: updating a state-transition model. *Ecosphere* 8:e01998
Read a discussion on this paper in [Decision Point #105](#)

- Pannell DJ & NA Glenn (2000). A framework for the economic evaluation and selection of sustainability indicators in agriculture. *Ecological Economics* 33:135-149.
- Pulliam, HR (1988). Sources, sinks, and population regulation. *American Naturalist* 132:652-661.
- Raiffa H & RO Schlaifer (1961). *Applied Statistical Decision Theory*. Graduate School of Business Administration, Harvard University, Cambridge, USA.
- Runge MC, SJ Converse & JE Lyons (2011). Which uncertainty? Using expert elicitation and expected value of information to design an adaptive program. *Biological Conservation* 144:1214-1223.
- Runting RK, KA Wilson & JR Rhodes (2013). Does more mean less? The value of information for conservation planning under sea level rise. *Global Change Biology* 19: 352-363.
Read a discussion on this paper in [Decision Point #67](#), p10-12.
- Walters CJ (1986). *Adaptive management of renewable resources*. MacMillan, New York, USA.
- Yokota F & KM Thompson (2004). Value of information literature analysis: A review of applications in health risk management. *Medical Decision Making* 24:287-298.



Learning about adaptive management

Good decisions produce results AND help you learn

By Michael Bode (Queensland University of Technology)

The world's ecosystems and species face a wide range of threats. Managers have a range of things they could try – 'interventions' – but they are often uncertain how the system will respond to any particular intervention. They are also often uncertain about which of the many threats is the most important to address. While additional research could allow us to work out which actions work and which may not, we don't have the time or the money to do all the research required for a 'perfectly' informed decision. We need to act now, in the face of all that we don't know.

Key messages:

Ecological problems are poorly understood, but often require immediate management action.

Adaptive management allows action to be taken immediately, and also helps learning.

Though commonly cited, true adaptive management is rarely applied.

Changing management because of ongoing monitoring at management sites is not adaptive management.

Adaptive management requires long funding time-frames, a high tolerance for risk, and institutional and management flexibility (factors that are rarely found in conservation).

Above: Fishery managers constantly have to deal with the challenge of making decisions on allowable catch quotas with incomplete information. Adaptive management is one pathway to meet this challenge. (Photo by Megan Saunders)

The harvesting of marine ecosystems (fishing) illustrates this dilemma well. Species targeted by commercial fisheries have both economic and conservation value. Managers have to regulate these fisheries to ensure the persistence of the harvested species, while also allowing vital economic activity to continue. The appropriate limitations on catch will depend on the dynamics of the ecosystem, but there are many unknowns. Managers are uncertain about how fast the population recovers from low abundance; about the strength of density-dependent feedbacks; about interactions between harvested species and other parts of the ecosystem.

Given enough time, research could resolve each of these unknowns. However, fishing – with or without regulations – will continue in the meantime. Managers are therefore faced with conundrum. Decisions need to be made immediately, but good decisions require additional information and a delay.

The theory of adaptive management, and the set of analytic tools that surround it, offer a path between these two common and contrasting demands. Adaptive management is often called 'learning by doing'. It recommends, as its name suggests, that managers act experimentally, undertaking the conservation action that they believe will simultaneously deliver benefits

and information. They watch the ecosystem respond to their actions, and then measure the consequences to better understand how the system operates – to ‘learn’ as they do.

However, adaptive management is a little more complicated than just trying something to see if it works. This approach is sometimes held up as adaptive management, and is often colloquially referred to as ‘suck-it-and-see’. This, however, is not a good representation of what adaptive management is about.

True adaptive management is active. It involves making an explicit prediction about how the system being managed will respond to a particular action, using multiple competing models of how the system works. And it ideally involves trialling multiple actions and comparing the results of these different ‘experiments’. It’s about learning about the system and then applying these lessons to subsequent management iterations, learning more with every iteration. It’s about undertaking a formal mathematical analysis of the problem in order to better understand how best to proceed.

The characteristics of adaptive management

The main aim of adaptive management in conservation is to identify and undertake actions that will deliver conservation benefits, but will also reduce our uncertainty about ecosystem dynamics. Management actions therefore have dual consequences, with explicit and implicit value to managers. Actions deliver **explicit** benefits when they achieve management goals. They also deliver **implicit** benefits when they help managers learn, because better information improves future decisions.

Achieving today’s management goals is obviously the most important, since it delivers immediate and direct benefits. Learning is ‘less’ valuable, since the benefits it offers will only occur in the future (and will therefore be time-discounted, see the box on ‘discounting active adaptive management’), and may be relatively small if the reduction of uncertainty is marginal. Nevertheless, their implicit value is real, can be quantified, and can change the optimal management action in significant ways.

Simple experimental management takes a passive stance towards learning. A manager undertakes the action that they believe is most likely to deliver benefits, and they then learn by observing the outcomes of this best-practice management. This is sometimes referred to as passive adaptive management.

Active adaptive management, by contrast, recommends a strategic approach to learning. The value of each action is

What is good ‘learning’?

Despite its focus on uncertainty and learning, adaptive management does not place an explicit value on learning. Gaining a better understanding of the ecological system, or of how it responds to management actions, is not a direct goal. Instead, lower uncertainty is valuable to the extent that it allows managers to better achieve the conservation goals.

Learning – with its attendant costs and delays – is never undertaken for its own sake and, as a consequence, adaptive managers don’t care what the largest uncertainties are. Instead, the analytic machinery of adaptive management incorporates a hidden value-of-information analysis. As with Value-of-information analysis, an adaptive manager would be happy to leave large sources of uncertainty unresolved, if the information would not change their decision. The result is a style of management that is highly (but specifically) tolerant of uncertainty.

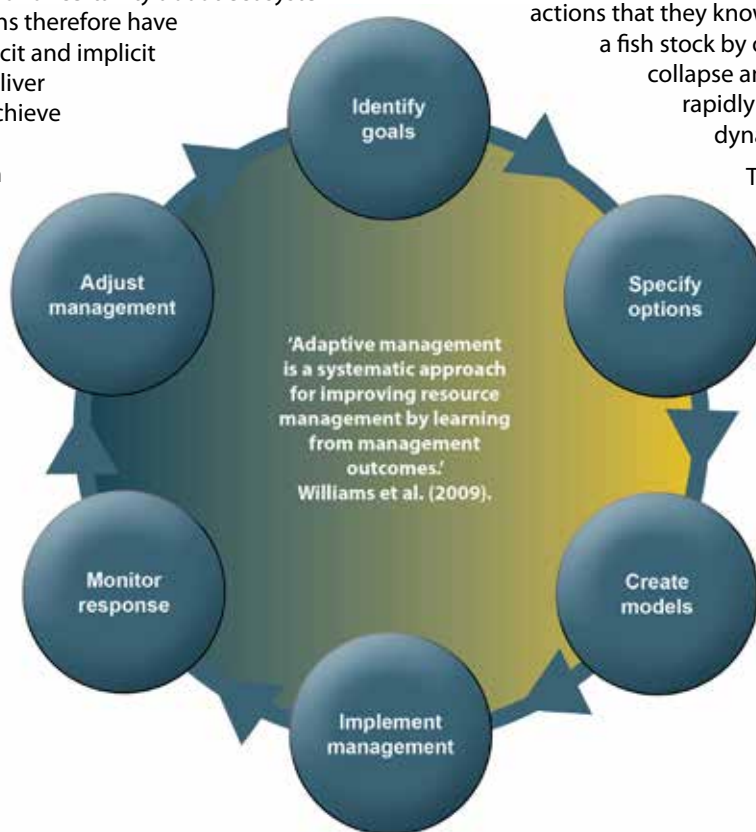
a combination of its expected beneficial outcomes, and also the future learning that could be gained. This combination makes adaptive managers more likely to choose actions that they actually think might be suboptimal, in the short term, to verify those beliefs. It can even encourage them to undertake actions that they know to be terrible, like collapsing a fish stock by overharvesting, if the resulting collapse and recovery will help them to rapidly learn about the ecosystem dynamics.

These facts can be distilled into a series of features that must be present for a project to be considered formal adaptive management. These are:


- 1. The identification of goals** for management (which are developed in collaboration with stakeholder groups).
- 2. The specification of multiple management actions** that could potentially achieve these management goals.
- 3. A dynamic model (or models) that predicts** how the system will change in response to management interventions, coupled with a statistical process for interpreting outcomes when they are observed. This model should be dynamic and stochastic, and the variation will be only partly predictable.
- 4. The implementation of**

at least one of the identified management actions, coupled with a monitoring program that observes how the system responds to the intervention.

5. A statistical updating of the system model(s) following the observed post-intervention responses, and a consequent change in management actions if this is recommended by the model.



A clear and succinct definition of adaptive management can be found in a guide on the topic published by the US Government (Williams et al, 2009): “Adaptive management is a systematic approach for improving resource management by learning from management outcomes.” “Learning” is important but in a decision framing that importance is determined by how much it improves the decisions we make. It is never undertaken for its own sake.



This whole process should be iterative, with reassessment of the system model and management intervention following monitoring, followed by a new phase of management.

Limitations to adaptive management

Adaptive management is a popular idea in conservation, and has been widely recommended in the applied science literature. Its popularity has reached the point where the term is routinely included in policy documents and legislation (eg, the Marine Life Protection Act in California). However, in the vast majority of cases, closer scrutiny reveals that these programs are not implementing adaptive management as described here.

Instead, decision-makers are using the term to describe management programs that are coupled with ongoing monitoring, and where the managers are uncertain about which intervention will be most effective. Most lack conceptual models of the system dynamics against which management outcomes can be compared. Thus, while they might be 'adaptive' in the broadest sense of the word (in that the management will 'adapt' as new information comes to hand), their approach to resolving management uncertainty is heuristic and informal. Consequently, these forms of pseudo-adaptive management are likely to be inefficient or ineffective. They will also definitely be suboptimal. They are better described as reactive management, or 'trial-and-error' management.

Adaptive management is often incorrectly applied because the term has become so broadly interpreted. However, even in those rare occasions where formal adaptive management has been applied, the approach has a low success rate. Reviews have attributed these failures to three primary issues:

1. inadequate or absent funding was made available for the monitoring step, which is essential for assessing the outcomes of actions;
2. decision-makers are often unwilling to admit their uncertainty, or to experimentally undertake management actions that they believe to be suboptimal; and
3. the scientific leadership, and the financial and political capital required to update a complex, adaptive program of management is rarely available on the necessary time-frames.

It should be noted that each of these issues is institutional, rather than technical. That is, the flaws are in the political science, not the biological science. In the latter case, researchers have pointed out an inherent contradiction in the implementation of adaptive management: the technical and scientific skills required to design and execute a complex experimental management plan are rarely found alongside the logistical and political acumen needed to execute such a plan in a difficult political environment.

Because none of these limitations are technical or scientific, they can't be addressed through better mathematical methods or decision-support tools. In broad terms, each of these constraints speaks to the inherent challenges of managing complex, stochastic ecosystems that evolve on timescales that are vastly longer than political – or even scientific – attention spans.

Adaptive management requires scientists and managers to transparently admit how little they understand about the systems they study. They must then seek funding to

The evidence on adaptive management

'Adaptive management' is everywhere. Google it and you'll get over five million hits, while academic search engines return over 20,000 articles. These articles discuss a huge range of topics – from ecology and conservation biology through to epidemiology, medicine and even construction. And more are being written all the time.

Adaptive management receives so much attention because it is intuitive, broadly applicable and conceptually appealing. Its basic premise is that as management proceeds, information is collected that improves knowledge of the system being managed. This knowledge is then used to improve future management practice.

Unfortunately, this simple overview masks a large amount of controversy in the academic literature (eg, see Keith et al, 2011). For example, some proponents have advocated that adaptive management be applied to nearly all environmental problems. Conversely, detractors have argued that adaptive management is merely a corporate buzzword, sometimes used to justify the continuation of flawed policies. Despite this diversity of opinion, quantitative evidence describing the pros and cons of adaptive management is difficult to find.

Working with David Lindenmayer and Gene Likens, Martin Westgate recently assessed the current 'state of the science' of adaptive management in ecology, with the aim of quantifying the usefulness of the concept (Westgate et al, 2013). They did this by finding articles whose authors stated that they were part of an adaptive management project, then looked for similarities and differences between these projects.

First, they quantified just how rare the actual application of adaptive management was in the academic literature. In a series of stages, they narrowed down their original 1,336 articles to 61 that explicitly claimed to enact adaptive management. Put differently, more than 95% of articles that discuss adaptive management don't test it.

Second, there is a notable lack of empiricism in the adaptive-management literature. The articles that we identified related to 54 separate projects, but most were only discussed in a qualitative manner, usually as a part of a broad review. Only 13 projects were supported by published monitoring data.

Finally, few projects were able to sustain the effort to complete the adaptive management cycle. Most papers that claimed to be starting adaptive-management projects were recent (within the last five years). Only four of the 13 adaptive management projects that we identified lasted longer than 10 years.

References

- Keith DA, TG Martin, E McDonald-Madden & C Walters (2011). Uncertainty and adaptive management for biodiversity conservation. *Biological Conservation* 144: 1175-1178.
- Westgate MJ, GE Likens & DB Lindenmayer (2013). Adaptive management of biological systems: a review. *Biological Conservation* 158: 128-139.
- For a discussion on this paper see [Decision Point #74](#)

Continued on page 39

Pushing the frontiers of AM

As any reader of *Decision Point* would know, over its life, CEED has explored and developed many aspects of adaptive management. Here are two recent examples.

Tracy Rout and Cindy Hauser and colleagues examined if adaptive management improved decisions surrounding searches for invasive species (Rout et al, 2017). They found that adaptive search strategies consistently outperformed alternative intuitive tactics. However, when they compared active and passive adaptive approaches to the searches they found there wasn't much difference in outcomes. As passive adaptive management is much easier to do than active forms it will often make sense to stick with passive.

Iadine Chadès and colleagues reviewed and updated what's known about the latest optimal or near-optimal approaches for solving adaptive management problems (Chadès et al, 2017). They reviewed three mathematical concepts required to solve adaptive management problems (Markov decision processes, sufficient statistics, and Bayes' theorem) and provided a decision tree to determine whether adaptive management is appropriate.

References

Chadès I, S Nicol, TM Rout, M Péron, Y Dujardin, J Pichancourt, A Hastings & CE Hauser (2017). Optimization methods to solve adaptive management problems. *Theoretical Ecology* 10: 1-20. <https://link.springer.com/article/10.1007%2Fs12080-016-0313-0>

Rout TM, CE Hauser, MA McCarthy & JL Moore (2017). Adaptive management improves decisions about where to search for invasive species. *Biological Conservation* 212: 249-255. <http://dx.doi.org/10.1016/j.biocon.2017.04.009>



When considering active adaptive management over short to medium time frames the optimal strategy may require the manager to be extra cautious. For example, in considering strategies for revegetation, we might actively experiment early on with different planting densities if we were in a position to consider the gain over a very long time frame. However, if we are only going to monitor the return over a medium time frame it's likely we won't get a decent return for our experimental effort. (See the box on discounting active adaptive management.) (Photo by David Salt)

Discounting active adaptive management

Discounting has a long history in the economic literature, and has found its way to ecology via fisheries and wildlife harvest management. The premise is that achieving my goals this year is more important to me than achieving my goals in some future year. For example, a fisher taking a 100 tonne catch this year can sell those fish and invest his profits at some interest rate. They expect that this investment will be worth more in 10 years time than the 100 tonnes of fish they'll catch and sell in that 10th year.

The future is also less valuable because it may never arrive. In 10 years, the fisher could be dead, or working as a chauffeur.

So, achieving our goals in the future isn't as valuable as achieving the same goals now. It follows that sacrificing our goals now for experimentation and learning may not be so worthwhile if it's only going to improve our achievements a long way into the future. Thus, discounting diminishes the value of experimentation, and experimentation is therefore less likely to be a part of the optimal active adaptive management plan.

It makes sense that experimental management may not be worthwhile when our time frame is short. If we experiment and jeopardise our goals at the beginning, we will probably need quite some time to learn about the system and make up for our losses with superior management based on better understanding. If it's important to achieve our goals over a short time frame then we are probably better off making do and acting on the limited information we already have, while still adjusting our actions if we see new clues along the way.

Until recently, prominent mathematical studies of active adaptive management have adopted long time frames, on the scale of decades or centuries. Mathematicians sometimes even determine the optimal management strategy over an infinite time frame! However, managers may also be faced with problems where there are only a handful of opportunities for monitoring, learning and adapting before the success of the project is evaluated.

Cindy Hauser and Hugh Possingham developed a model of adaptive management and investigated the merits of experimental actions over a variety of time frames. What they discovered was quite a surprise. Their model was consistent with prevailing thought over long time frames – when there's substantial uncertainty about the best way to manage our ecosystem, some early experimentation is beneficial. However, over medium-length time frames (such as 10-15 years), the optimal active adaptive strategy may be extra cautious. Actions that could cause ecosystem damage are avoided, even if that risk is acceptable under the 'balance of current evidence' passive adaptive strategy.

Over medium time frames, there is insufficient time to learn, adapt and recover any losses incurred while learning. An action that is reliable, albeit probably inferior, is preferable. Thus, the active adaptive manager may be a model of caution rather than daring experimentation.

Reference

Hauser CE & HP Possingham (2008). Experimental or precautionary? Adaptive management over a range of time horizons. *Journal of Applied Ecology* 45: 72–81. And see a discussion of this paper in [Decision Point #18](#), p4,5.

An illustrative example

In its application, adaptive management is a process of dynamical optimisation under uncertainty. Its mathematical formulation explicitly describes the uncertainty associated with key system parameters. The problem dynamics include (1) a model of how each action is expected to change the objective function, and (2) a model of how observations of the outcome alter our understanding of the system dynamics – how we learn. Dynamical optimisation tools (eg, SDP, optimal control) are used to consider sequences of decisions, looking forward in time.

These dynamics are most easily illustrated with a simple example. Imagine a manager is attempting to conserve a threatened species. Two management actions are available to her. The first action is the current best-practice, and its probability of success is known with complete confidence. The alternative action has never been applied before, and so it is unclear whether it is better or worse than the current best-practice action. If successful, each action would deliver the same benefit to the species, and the only issue is therefore whether the new action is better or worse than the current best-practice.

If the manager were only making a single, one-off management decision, then the appropriate choice would be to implement the best-practice action, since it has a demonstrated track-record of success and, without subsequent actions, there is no value to learning about the alternative.

However, if she intends to continue managing for multiple time-steps, there may be value in exploring the alternative action, since there is some probability (shaded in red in Fig 1) that the alternative action is superior. We have therefore set up a simple adaptive management problem. Managers have two actions available to them. They need to take actions immediately, but they don't know everything they would like to know about the system – specifically, whether the alternative management action is better or worse than the best-practice action. They therefore consider taking actions that will offer both information and benefit. Below, we explore these ideas by formulating and solving this problem as a simple, two-step adaptive management project.

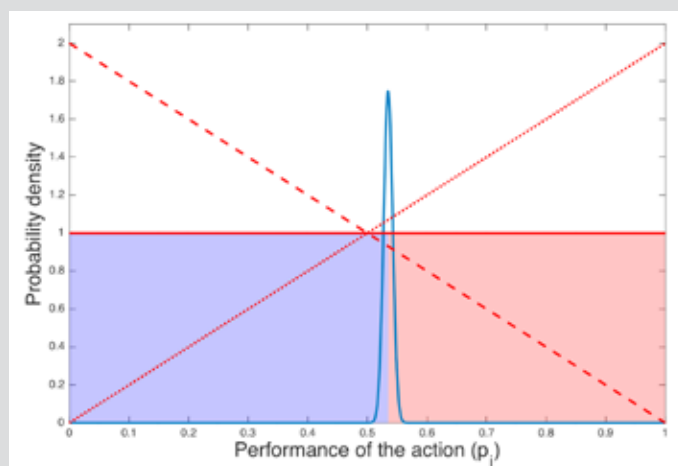


Fig 1: Belief distributions of the adaptive manager in our simple example. The solid blue line indicates that the manager's belief distribution in the success rate of the best-practice action centres around 0.53, and has very low uncertainty. The solid red line indicates the manager's deep uncertainty about the performance of the new, alternative action. The shaded blue region is the probability that the best-practice action is superior; the shaded red region is the probability that the alternative action is better. The two other red lines show what the manager's belief distribution would look like if her first attempt to apply the alternative action yielded a success (dotted line, showing a stronger belief in its success), or a failure (dashed line, reflecting the evidence of its failure).

In the first time-step, the manager could undertake the best-practice action, and we can easily calculate the expected value of this choice. In time-step 1, this action will deliver a known expected benefit of p_1 (blue line in Fig 1), since the probability of success is known with (almost complete) certainty. The manager will then make the same decision in the second time-step: with no subsequent actions there is no benefit to exploring the alternative action, and the optimal choice will therefore be to again undertake the best-practice action. Starting with the best-practice action therefore has an expected benefit of $B_1 = 2p_1$ (without time-discounting).

Alternatively, the manager could undertake the alternative action in the first time-step, observe the outcome, and learn about its success rate before making her second decision. Given we've never seen the alternative in action, we start with a uniform belief in its success rate. Perhaps it is always successful ($p_2 = 1$); perhaps it will never work ($p_2 = 0$). We can capture this belief with a uniform distribution, which we describe using the beta function $f(p_2) = B(1,1)$ (the solid red line in Fig 1).

In the first time-step, the expected benefit of the alternative action will reflect the manager's informed-but-uncertain belief distribution in the method's success rate. Given our belief distribution, this probability is $p_2 = 1/2$. The manager's actions in the second time-step will reflect the outcome of this first attempt.

On the one hand, if the action was unsuccessful (with probability $1 - p_2 = 1/2$), she will have learned a pessimistic lesson about the alternative action. Specifically, her new belief about the action can be described by the beta distribution $B(1,2)$ (the dashed red line in Fig 1). From the properties of the beta distribution, the expected benefit of taking the alternative action in the second time-step would then be $p_2 = 1/3$. On the other hand, if the action was successful, the expected benefit of taking the alternative action in the second time-step would be $p_2 = 2/3$ (dotted red line in Fig 1).

We note first that, if the probability of the best-practice action is lower than 1/3 then the manager will automatically take the alternative action in both time-steps, since the expected outcome will always be higher. Second, we note that if the best practice

Continued next page 39

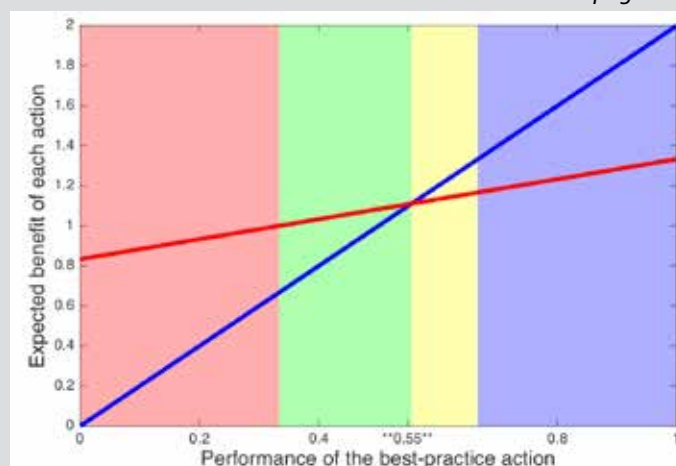


Figure 2: Optimal adaptive management approach to the simple management problem. The x-axis shows the manager's belief in the performance of the best-practice action (the value of 0.55 reflects the situation in Fig 1). The blue line shows B_1 , the benefit that the manager can expect if she starts by applying the best-practice action in the first timestep. The red line shows B_2 , the benefit she can expect if she starts with the alternative action. The optimal adaptive management decision for a given belief in the best-practice action is determined by the higher of the two lines.

Continued from page 38

action has a probability of success higher than $2/3$, there is no point to learning about the alternative action, since even a successful application would still leave the best-practice action a superior choice. Learning in this case cannot alter the management decision, and there is therefore no value to this information.

Following this adaptive approach, we can predict that the expected benefit of taking the alternative action in the first timestep is:

$$B_2 = p_2 + p_2 p_2 + (1 - p_2) p_1 = 1.083$$

This equation implies that an adaptive manager faces 1 of 5 scenarios (Fig 2). They are ordered here from low p (the best practice action is awful) to high p (it usually succeeds)

$p_1 < 1/3$ (red shaded area in Fig 2): In this case, the manager will undertake the alternative action in both time-steps. Even if the alternative action fails in the first application, it is still expected to outperform the best-practice action.

$p_1 > 1/3$ (blue shaded area in Fig 2): In this case, the manager will never attempt the alternative action. Even if it were successful in the first application, its expected performance will still be lower than the best-practice action. Note that in this case, we are aware of the substantial probability that the alternative action is superior to the best-practice action. However, the adaptive management analysis indicates that there are too few learning opportunities for management to resolve the question.

$1/3 \leq p_1 < 1/2$ (green shaded area in Fig 2): In this case, the manager suspects that the alternative action will be the better option, but will keep an open mind. In the first time-step they take the alternative action, and make their second decision after observing its performance.

$1/2 \leq p_1 < 0.555$ (green shaded area in Fig 2): Based on the current information, the manager believes that the best-practice action is superior to the alternative action. Nonetheless, they will undertake the alternative action in the first time-step, because the value of learning about its true performance outweighs the short-term loss of expected benefits.

$0.555 < p_1 < 2/3$ (yellow shaded area in Fig 2): In this situation, the manager is sufficiently confident about the superior performance of the best-practice action that they are not willing to learn about the alternative action. This is despite the very real possibility (perhaps as high as 45%) that the alternative action is superior. The value of obtaining this information is just not high enough to justify its short-term costs.

This problem is a conservation realisation of the 'one-armed bandit' problem, first solved by economists in the 1950s, and famous for its application to clinical trials of medical interventions. There are a range of methods available for calculating or approximating the optimal solution in more complex contexts, and all agree that the best way forward is a mixture of exploitation and exploration. Exploitation involves the application of the action that we currently believe to be superior (action 1, in Fig 1); exploration requires the judicious use of uncertain actions that may (or may not) turn out to be superior than the current best practice (action 2; Figure 1).

The optimal balance of exploitation and exploration will depend on a suite of factors, including the amount of uncertainty associated with each action, the length of the management horizon (ie, are we planning to manage for 2 years, or 200 years?), and the broader applicability of the actions.

experimentally manage these systems over very long timescales, during which time they will have to consistently cite their own ignorance as a rationale for (potentially unpalatable) experimental management. Finally, they are required to maintain attention and focus on this problem for long periods, during which time they may be able to draw few useful conclusions.

Successful applications of adaptive management

Despite these constraints, adaptive management can be successful, even at extensive spatial and temporal scales, and in socio-ecological systems that include large numbers of stakeholders with disparate values and goals. As evidence, there are a small number of well-documented examples where the application of formal adaptive management has delivered management benefits.

The best-known example is of waterfowl management in the United States, which has been managed with a formal adaptive management program for the past three decades. Other examples include the management of Glen Canyon, on the Colorado River, USA, and the restoration of sand-mined ecosystems in Australia.

Although each of these success stories applied the same formal adaptive management techniques, each case-study is also deeply idiosyncratic. In each example, managers found unique ways to address a series of difficult social, fiscal and political constraints. The examples therefore provide lessons, but not necessarily broad solutions, to the factors that are considered the greatest obstacles to successful adaptive management.

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Recommended reading

- Hughes TP et al (2007). Adaptive management of the Great Barrier Reef and the Grand Canyon world heritage areas.
AMBIO: A Journal of the Human Environment 36: 586-592.
- Lyons JE et al (2008). Monitoring in the context of structured decision-making and adaptive management.
Journal of Wildlife Management 72: 1683-1692.
- McCarthy MA & HP Possingham (2007). Active adaptive management for conservation.
Conservation Biology 21: 956-963.
- Rumpff L et al (2011). State-and-transition modelling for adaptive management of native woodlands.
Biological Conservation 144: 1224-1236.
- Walters CJ & R Hilborn (1978). Ecological optimization and adaptive management.
Annual review of Ecology and Systematics 9: 157-188.
- Walters CJ (1986). *Adaptive management of renewable resources*. Macmillan Publishers
- Walters CJ (2007). Is adaptive management helping to solve fisheries problems?
AMBIO: A Journal of the Human Environment 36: 304-307.
- Williams BK, RC Szaro & CD Shapiro (2009). *Adaptive Management: The U.S. Department of the Interior Technical Guide*. Adaptive Management Working Group, US Department of the Interior, Washington DC.



The importance of long-term monitoring

Good decisions for the environment need an eye on the longer term

By David Lindenmayer (The Australian National University)

Effective long-term environmental monitoring is difficult and challenging; it requires good design, careful review, long-term commitment, and often gets overlooked when resources are handed out by our political leaders. Given this, why bother? We bother because long-term monitoring is the cornerstone of effective environmental policy and management. In a 'post-truth' age witnessing a crisis in biodiversity decline, long-term monitoring is something we can't afford not to do.

However, if you are going to do it, it needs to be done properly. Literally thousands of scientific articles and dozens of books have been written on almost all aspects of long-term monitoring. This chapter does not attempt to repeat or even briefly summarise this vast body of work (though for a good guide to it see Lindenmayer & Likens, 2018). Rather, here we focus on five key learnings from the body of work on long-term monitoring as they relate to making good environmental decisions.

1. Evidence-based policy needs long-term monitoring

The mantra of modern governments and other bodies responsible for managing natural resources (including biodiversity) is that both management and policy must be 'evidence based'. In a world in which 'truth' is constantly under attack the need is only greater, but where does that evidence come from. Long-term monitoring is often the essential source.

(Above) Billions of dollars have been invested in large-scale restoration programs across farming landscapes in Australia and overseas. Some projects involve the protection of remnant native vegetation, others involve linear or block plantings of native trees. Some involve innovative mixes of native and traditional crops. Which approaches work? Which designs are most cost effective and enduring? Long-term monitoring can generate the evidence on which to judge these programs and build better policy (evidence-based policy). Unfortunately, long-term monitoring for such programs is more the exception than the rule. (Photo by Dean Ansell)

Key messages:

1. Long-term monitoring provides essential evidence on which to base good environmental decisions.
2. Good design is essential for effective long-term monitoring.
3. Things change over time; to remain effective, long-term monitoring needs to adapt around these changes.
4. Partnerships are crucial for ensuring long-term monitoring is maintained and listened to.
5. Long-term monitoring is most effective where it is complemented by other value frames (such as economics).



Part of the long-term monitoring program established for the Environmental Stewardship Program. (Photo by David Salt)

In terms of biodiversity, long-term monitoring is often needed to measure change in a given entity (such as a population of a species or the condition of an ecosystem), but also to measure how those entities change in response to some kind of management intervention (like pest control or habitat enhancement). Long-term monitoring is essential to determine if actions taken to manage the environment are effective, and therefore whether decisions made to invest in particular actions are vindicated (or whether different interventions are needed).

Consider the case of controlling Bitou bush, one of Australia's worst invasive plant species (ironically, it was deliberately introduced in the first instance to help with sand-dune stabilisation). Long-term monitoring enabled us to determine the effectiveness of a program to control the Bitou bush in Booderee National Park in southern New South Wales. A significant proportion of the management budget for the park is dedicated to controlling this weed so dealing with it comes with a high opportunity cost (in terms of other work not being funded).

Despite the considerable expense, prior to us monitoring efforts at control, it wasn't known whether the program was

What is long-term monitoring?

There are many formal definitions of what constitutes long-term monitoring but a good rule of thumb I apply is that it is any investigation involving repeat measurement that has been running continuously for ten or more years. Ten years is not a magical number separating 'short-term' from 'long-term', however monitoring programs that have run for longer than ten years usually have a 'long-term' framing aimed at capturing trends and variability that are often not evident in shorter programs.

either ecologically effective or cost effective. Our monitoring work, carried out in collaboration with Booderee National Park management (see the box '15 years at Booderee'), revealed that the currently applied method of spraying Bitou bush, then burning the dead canes, following by respraying was the most ecologically effective treatment protocol for removing the invasive bush. Not only did it control the weed, this treatment also stimulated the restoration of native plant cover (Lindenmayer et al, 2015c).

The spray-burn-spray protocol was also found to be more cost-effective than other kinds of management interventions. Moreover, very few native animal species were found to be disadvantaged by the spray-burn-spray treatment protocol (Lindenmayer et al, 2017). This is an important outcome as Booderee National Park is a stronghold for populations of endangered species such as the eastern bristlebird. Such taxa were found to benefit from the removal of Bitou bush and post-treatment recovery of native vegetation (Lindenmayer et al, 2017).

The problems of not conducting long-term monitoring are evident from many failed environmental programs, including those in which very large investments were made.

For example, despite billions of dollars of investment in river restoration programs in the USA, a paucity of robust long-term monitoring made it impossible to determine whether such restoration efforts had been effective and successful (Bernhardt et al, 2005).

This is a far from isolated case. The effectiveness of billion dollar agri-environment schemes to better manage biodiversity and other conservation values in farming landscapes in Europe and North America is poorly known because of a lack of long-term monitoring. Similarly, large scale vegetation restoration and salinity mitigation programs funded by the Australian and State Governments remain poorly monitored (if monitored at all). This fundamental oversight leads to ineffective programs, vast amounts of wasted taxpayer funding and a public misperception that environmental problems cannot be resolved (Hajkowicz 2009).

Policy makers and politicians can certainly make poor environmental decisions, even in the face of overwhelming evidence of the need for alternative decisions to the ones they have made. We trust, however, that such poor decisions will be rarer when evidence is available than when there is no available evidence. Put another way, good evidence, based on long-term monitoring should provide scientists with the persuasive power to influence decisions, and even if not an 'optimal' decision, then perhaps 'less bad' than it might otherwise have been.

Adaptive monitoring and the ESP

The Environmental Stewardship Scheme (ESP) was established to assess the effectiveness of management interventions associated with a major agri-environment scheme in the temperate woodlands of eastern Australia. Farmers are paid to carry out specific management actions to improve the condition of patches of endangered Box Gum Grassy Woodlands (BGGWs) on their land. The ESP comprises a total of 158 farms in a region stretching over 2000 km (south to north) (Burns et al, 2016). A patch of BGGW targeted for stewardship management on each of the 158 farms is also targeted for monitoring with a matched control patch (where no stewardship management occurs) also monitored on each farm.



A monitoring station at one of the Environmental Stewardship Program sites in a grassy woodland on private land. Tiles, rotting sleepers and sheets of tin provide refuge used by a range of native animals. (Photo by David Salt)



Signage on the fence surrounding an ESP site. (Photo by David Salt)

The budget for the monitoring was initially substantial as the government agency responsible for the ESP demanded that every patch on every farm be monitored every 1-2 years. However, funding cuts occurred four years into the 15-year program.

Adaptive monitoring had to be adopted to prevent the entire monitoring effort collapsing. A rotating sampling monitoring approach was introduced in which 65% of farms were surveyed in any given year with a complete 'census' of all farms undertaken twice in four years (Lindenmayer et al, 2012).

The emphasis of monitoring switched from an assessment of compliance for implementing particular kinds of conservation management to an estimation of changes in condition across a large 'population' of sites.

Unfortunately, even this adaptive change was insufficient to enable a long term assessment of the effectiveness of the scheme because, subsequent to this, the Australian Government changed the monitoring approach using a different group and methodology. This meant that what had been done before does not match with the limited effort that is being done now, thereby breaching the integrity of the time-series data gathered over the past 8 years and precluding robust assessment of whether the programme is effective (or otherwise) (Lindenmayer & Likens, 2018; Lindenmayer et al, 2018). All of which highlights the many challenges of running an effective long-term monitoring program.

2. Effective long-term monitoring is built on good design

Long-term monitoring programs need to be underpinned by good design if they are to generate data that can guide effective environmental decisions. And that design begins with asking what the monitoring will actually be used for. And, if there is no intention on the part of managers to change their management or if there is no capacity to learn from the monitoring results, then a monitoring program may not even be appropriate (see [Decision Point #52](#) p4-7).

If there is capacity to learn and a willingness of managers to respond, then there are some fundamental ingredients which contribute to good monitoring design. These include:

- a. Careful articulation of the objectives** of monitoring, with all partners being clear about the aims and objectives;
- b. Good and tractable questions of management relevance** (often being informed by a well-developed conceptual model of the system being monitored);
- c. Implementation of a robust statistical design** (that answers key questions);
- d. Regular assessment of the data gathered** (to ensure errors in a dataset are corrected or key missing variables can be gathered); and
- e. The inclusion of trigger points** for action if major changes occur in the system being monitored (Lindenmayer et al, 2013b).

Conversely, long-term monitoring programs established without these considerations can result in an expensive waste of resources. An example is the Alberta Monitoring Program (Alberta Biodiversity Monitoring Institute 2009) in the Canadian Province of Alberta in which millions of dollars are expended each year on a passive monitoring program that lacks questions or a robust underlying statistical design (see Lindenmayer & Likens, 2010a).

3. Adaptive monitoring can be essential

Things change, it's a given. It's better to adapt to changes than stick with a monitoring program that is no longer relevant.

Often there is a need to change the questions being posed over time and/or change the underlying experimental design in response to those changed questions. Or there might be other reasons for change like the development of new technology that requires altered field-based measurement protocols.

Poor earlier policy and/or management decisions also might create extra cascading environmental problems, demanding a reworking of the original scope of a monitoring program. An adaptive monitoring approach (as described in Lindenmayer and Likens 2009) may be required to redesign a pre-existing monitoring program so that it can answer new key questions of management relevance that are useful in guiding environmental decisions.

An example of adaptive monitoring comes from monitoring the Environmental Stewardship Program (see the box on 'Adaptive monitoring and the ESP').

4. Partnerships are critical

Effective long-term monitoring programs need good partners – partners that will help frame the purpose of the program, assist in translating the monitoring results into effective management decisions, and act as champions for the program to ensure it has a long-term future.

15 years at Booderee

Strong and enduring partnerships have been at the heart of the success of the 15-year monitoring program at Booderee National Park (Lindenmayer et al, 2013a) on the south coast of NSW. The data from the monitoring program have underpinned approaches to fire management by the resource managers of the park.



David Lindenmayer (on the left) explains the monitoring program at Booderee to a group of fire ecologists. In front of them is a set of pit fall traps that form part of the program. (Photo by David Salt)



Booderee National Park Resource Manager Nick Dexter (foreground) discusses Bitou bush control with scientists and managers during a science workshop in the park. The science/management relationship that has been cultivated at Booderee has made an important contribution to conservation in the coastal reserve. (Photo by David Salt)

The partnership has focussed on three key issues within Booderee: the impacts of fire on native biota, the response of vertebrates to feral animal control and the control of Bitou bush.

In regards to fire, a new understanding of the relationships between bird persistence and recovery following fire (derived from empirical research) has resulted in a change from uniform prescribed burning of entire compartments of native vegetation to patchy fires across a maximum proportion of a given compartment.

Monitoring has also demonstrated the value of feral animal control showing it substantially increases populations of some animals such as the common brushtail possum, the long-nosed bandicoot and the endangered eastern bristlebird. On this basis, an intensified approach to feral animal control in Booderee National Park is now well established as a key and ongoing conservation activity recognised formally within the official management plan for the reserve.

Partnerships between scientists, resource managers and policy makers can ensure that the key questions being addressed in a long-term monitoring program are management relevant but at the same time scientifically tractable. Partnerships also provide a vehicle for regular exchange of information and the opportunity to build a broad constituency to maintain long-term work. Such partnerships are essential to ensure that the evidence gathered from long-term monitoring can be widely communicated to those responsible for decision making; this may include engagement with the political process to inform ministers (and minister's advisors) on what the results of long-term monitoring are showing.

Considerable effort is needed to maintain the array of partnerships which underpin long-term monitoring and its link with effective environmental decision making. For example, the rapid turnover (churn) of staff within government agencies poses a particular challenge as champions for particular projects are needed to maintain them in the long term.

Considerable time often needs to be expended by the scientific leader of a long-term project to explain what the work aims to do, why it is important, why it is relevant to informed policy and decision making. This may need to be done repeatedly as new staff are recruited. Field trips to long-term monitoring sites can sometimes be particularly effective as these provide a practical and tangible context for how particular management problems are being examined and tackled through science-manager partnerships (Lindenmayer et al, 2013a).

5. But remember, 'it's the economy, stupid!'

Many long-term monitoring programs focus on threatened species and ecosystems and we know from experience this is a good basis for deciding how to effectively manage these systems. However, when it comes to our political representatives, long-term biophysical evidence is often of secondary significance in the political calculus. They are more interested in what it means for their voters which is why when considering the outputs of long-term monitoring programs it's always valuable to consider how they can be integrated with other metrics relating to your system of interest. The environment is important but the social and economic dimensions of the system are possibly of greater significance when it comes to policy and decision making.

As an example, much has been written about the results of long-term ecological and environmental monitoring in the montane ash forests of the Central Highlands of Victoria (home to the Critically Endangered Leadbeater's possum and several other threatened species; (Lindenmayer et al, 2015a). Unfortunately, much of the conservation science generated over many years remains ignored (Lindenmayer et al, 2015b). However, monitoring may have more traction with decision makers when key natural assets are monetized in economic and environmental frameworks like those developed by the United Nations, for example the System of Economic and Environmental Accounting (or SEEA).

The SEEA framework enables the 'value-added value' of industries based on natural resources like tourism, carbon, water and timber to be compared in a formal and internationally accepted accounting framework. When applied to the forests of the Central Highlands it showed that the value-added value of the native forest timber industry (\$12 million)

was a fraction of the water industry (\$310 million) and the tourism sector (\$260m) (Keith et al, 2017). Decisions to maintain timber production (which undermines the value of the water and tourism industries) are therefore based on something other than rational economics.

Notably, in a communique from a 2016 COAG meeting, the Commonwealth Minister for the Environment and Energy and his State and Territory colleagues recommended that environmental accounting be widely applied and adopted in Australia. We suggest that the approach has the potential to add considerable value to datasets that are being gathered in environmental monitoring programs and provides a new way that such programs can help influence decision making.

Decisions and long-term monitoring

Long-term monitoring programs are often linked with many kinds of decisions; some associated with better informing on-the-ground management, others linked with changes in policies. There are also scientific decisions associated with the 'inner workings' of long-term monitoring programs such as the way they are designed or re-designed and how protocols for field measurements might be altered on the basis of the development of new techniques or the discovery of new problems (such as the colonization of new species of invasive organisms).

The five major themes discussed here can potentially influence each of these kinds of decisions and vice-versa. That is, management, policy and scientific decisions can influence (and also be influenced by) the need to make good environmental decisions, the importance of good study design in monitoring, the need for adaptive management, the fundamental importance of partnerships, and recognition of the potential value of extending biodiversity monitoring into other domains.

Regardless of what influences what, the case for good long-term monitoring as an (evidence) base for better decision making is indisputable.

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References and further reading

- Alberta Biodiversity Monitoring Institute (2009). *Program overview*. Alberta Biodiversity Monitoring Institute, Edmonton.
- Bernhardt ES, MA Palmer & JD Allan (2005). Synthesizing US river restoration projects. *Science* 308: 636-637.
- Hajkowicz S (2009). The evolution of Australia's natural resource management programs: Towards improved targeting and evaluation of investments. *Land Use Policy* 26:471-478.
- Kay G, PS Barton, D Driscoll, S Cunningham, W Blanchard, S McIntyre & DB Lindenmayer (2016). Incorporating regional-scale ecological knowledge to improve the effectiveness of large-scale conservation programmes. *Animal Conservation* 10: 515-525.
- Keith H, M Vardon, J Stein, J Stein & DB Lindenmayer (2017). Ecosystem accounts define explicit and spatial trade-offs for managing natural resources. *Nature Ecology and Evolution* 1: 1683-1692.
- Lindenmayer DB & GE Likens (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology and Evolution* 24:482-486.

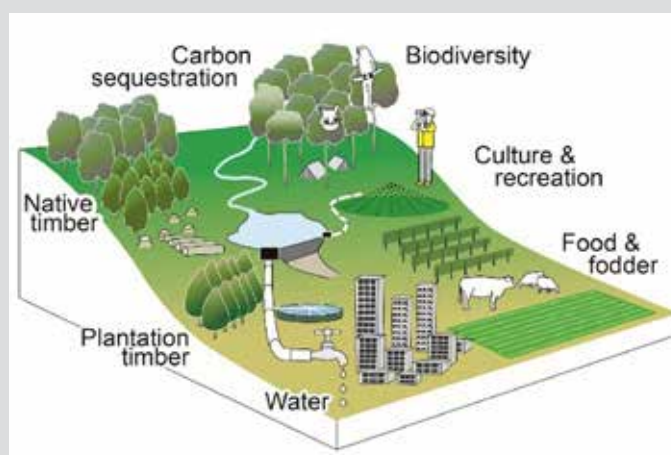
Future challenges

Despite the extensive work completed on long-term monitoring and how it might be better designed and implemented, some important challenges remain. First, monitoring programs need to better account for cumulative effects of multiple, interacting disturbances in ecosystems and, for example, help us better predict major problems such as the risk of ecosystem collapse (Sato and Lindenmayer 2018).

In addition, more information is needed on the effectiveness of monitoring in terms of return on financial and logistical investment to help better answer questions like: What is the value of new information gained from monitoring? and How does that information help contribute to better decision making? New insights on return on investment are also critical for determining when it is time to stop monitoring as well as when not to start new monitoring programs (see also (McDonald-Madden et al, 2010).

Finally, there is a need to revisit the already large literature on the connections between science and policy and, in a post-truth era (Lubchenko 2017). We need to explore:

- (1) How to reinstate the value of empirical evidence in making better decisions and
- (2) How can we reduce the amount of time between gathering and analysing data in monitoring programs and making better decisions



Long-term biodiversity monitoring data enrich our understanding of the whole-of-landscape context for environmental accounting. This is one way in which greater connections can be made between science and policy.

Lindenmayer DB & GE Likens (2010). The science and application of ecological monitoring. *Biological Conservation* 143:1317-1328.

Lindenmayer DB, C Zammit, SA Attwood, E Burns, CL Shepherd, GE Kay & J Wood (2012). A novel and cost-effective monitoring approach for outcomes in an Australian biodiversity conservation incentive program. *PLOS One* 7:e50872.

Lindenmayer DB, C MacGregor, N Dexter, M Fortescue & P Cochrane (2013a). Booderee National Park management: Connecting science and management. *Ecological Management & Restoration* 14:2-10.

Lindenmayer DB, M Piggott & B Wintle (2013b). Counting the books while the library burns: Why conservation monitoring programs need a plan for action. *Frontiers in Ecology and the Environment* 11:549-555.

Lindenmayer DB, D Blair, L McBurney & S Banks (2015a). *Mountain Ash. Fire, Logging and the Future of Victoria's Giant Forests*. CSIRO Publishing, Melbourne.

Lindenmayer DB, D Blair, L McBurney & SC Banks (2015b). Ignoring the science in failing to conserve a faunal icon – major political, policy and management problems in preventing the extinction of Leadbeater's possum. *Pacific Conservation Biology* 21:257-265.

Lindenmayer DB, J Wood, C MacGregor, Y Buckley, N Dexter, M Fortescue, RJ Hobbs & J Catford (2015c). A long-term experimental case study of the ecological effectiveness and cost effectiveness of invasive plant management in achieving conservation goals; Bitou Bush control in Booderee National Park in eastern Australia. *PLOS One* 10:e0128482.

Lindenmayer DB, J Wood, C MacGregor, RJ Hobbs & J Catford (2017). Non-target impacts of weed control on birds, mammals and reptiles. *Ecosphere* 8: <https://doi.org/10.1002/ecs2.1804>

Lindenmayer DB & GE Likens (2018). *Effective Ecological Monitoring*. (Second Edition) CSIRO Publishing, Melbourne.

Lindenmayer DB, D Michael, M Crane & D Florance (2018). Ten lessons in 20 years: Insights from monitoring fauna and temperate woodland revegetation. *Ecol Manag & Restor* 19: 36-43.

Lubchenko J (2017). Environmental science in a post-truth world. *Frontiers in Ecology and Environment* 15:3.

McDonald-Madden E, PWJ Baxter, RA Fuller, TG Martin, ET Game, J Montambault & HP Possingham (2010). Monitoring does not always count. *Trends in Ecology and Evolution* 25:547-550.

Sato C & DB Lindenmayer (2018). Meeting the global ecosystem collapse challenge. *Conservation Letters* 11 <https://doi.org/10.1111/conl.12348>

Sato C, J Wood, JA Stein, M Crane, S Okada, D Michael, G Kay, D Florance, J Seddon, P Gibbons & DB Lindenmayer (2016). Natural tree regeneration in agricultural landscapes: The implications of intensification. *Agriculture, Ecosystems & Environment* 230:98-104.

United Nations (2012). *System of Environmental-Economic Accounting Central Framework*. United Nations, New York.

Welsh AH, RB Cunningham & RL Chambers (2000). Methodology for estimating the abundance of rare animals: seabird nesting on North East Herald Cay. *Biometrics* 56:22-30.



Decision science and environmental policy

Achieving better outcomes from environmental policy is more than just good science

By David Pannell (University of Western Australia)

The creation and design of an environmental policy involves a myriad of decisions. Here are just a few of them:

What's the need?

Should there be a policy at all? Which specific environmental issues should be covered by the policy? What goals or targets should be set for the policy? Over what spatial scale should the policy operate? Over what time frame should it operate?

Key messages:

1. Decision science is only one of many factors that may be relevant in the environmental-policy process.
2. Decision science has enormous potential to make policies more effective at delivering outcomes that are highly valued by the community.
3. There are a number of challenges for decision science if it is to meet its potential.
4. There are many examples where decision science has met these challenges and made important contributions to environmental policy.
5. Key success factors for decision scientists to contribute to environmental policy include good communication, pragmatism, patience, persistence, a capacity to develop meaningful relationships with policy makers, and choosing the right decision-science projects in the first place – ones that are likely to make a difference.

Above: Decision scientist and policy person out in the field. Researchers want their research to inform policy and management but it often doesn't happen. For greater interaction, scientists need to acknowledge the many cultural differences between the the world of research and policy formation. (Photo by David Salt)

Where's the emphasis?

Should effort and funding be targeted to key priority issues or locations, or spread relatively evenly across issues or locations?

What's the approach?

Should the policy involve public education? If so, who needs educating, what about, and how? Should the policy involve regulation of the actions of individuals or businesses? If so, how large should the fines for non-compliance be? How much should be invested in inspection and testing of compliance?

Should the policy involve payments to reward positive environmental behaviour? If so, how large and how frequent should the payments be? Should people be rewarded for approved actions, or only if those actions result in positive environmental changes? Is research or a pilot study needed prior to full implementation? What approach to monitoring the results of the policy should be used?

And, of course, each question (of which the above is just a logical beginning) spawns its own set of sub-questions, which in turn generate even more, and so on.

The relevance of decision science

While easy to pose, most of these questions are not straightforward to answer. The alternative options to each decision may have multiple consequences, some of which are

hard to predict. In most cases, there is high uncertainty about the environmental consequences of the various policy options. Relevant considerations include social, administrative, political and financial issues, not just environmental consequences. These different issues need to be drawn together and weighed up, but the best way to do that may not be apparent.

Decision science is definitely relevant to this. It provides a set of concepts, models and tools that have been developed to help address complex decisions under uncertainty. It aims to formulate each decision problem into a coherent and logical structure, so that it is easier to think about the options. In so doing, the consequences of these options become more transparent and easier to compare.

This helps decision makers compare and rank policy options in ways that are logically sound. While this seems like a sensible and rational thing to do, it's something that environmental managers and policy makers often fail to practice.

For example, we know a lot about what are good approaches to prioritisation and yet this knowledge is frequently not applied (eg, consider how prioritisation metrics are often poorly constructed, see [Decision Point #82](#)). Or, as another example, a lot of decision science has gone into how biodiversity offsets should be estimated (see [Decision Point #69](#), p10,11) but these lessons are sometimes not well accounted for in policy.

If done well, decision science should also help decision makers avoid some of the biases that plague most decision making (see the box on bias and NRM and [Decision Point #93](#)). On the face of it, decision science should be highly useful to environmental policy makers.

Nevertheless, for a range of reasons (some of which are discussed here), most environmental policy decisions are made without help from decision science. There are many excellent examples where it has played a helpful role, or even a pivotal role, but they are the exceptions. The more common situation is that prioritisations are done poorly or not at all, that decisions are made without thorough and systematic analysis of the options, and that ongoing monitoring of the outcomes of our environmental decisions is done inadequately.

In this article I will reflect on the nexus between environmental policy and decision science. What aspects of the policy process make it difficult for decision science to play a bigger role? I'll briefly present some examples of environmental policy development with varying levels of use of tools developed by decision scientists. And, for those decision scientists who aspire to play a bigger role in environmental policy, I'll discuss strategies they may want to consider.

Challenges for decision scientists

In my time working in the field of environmental decision science, working with researchers from a variety of disciplines, I have identified a range of challenges that have increased the difficulty of decision science influencing policy outcomes. These include the following:

Non-scientific considerations matter: Sometimes scientific information may be known to policy makers, but the decision reached may still appear to be inconsistent with the science. This may or may not be a concern. It may simply reflect that policy makers and managers must consider additional factors, such as legal mandates, societal desires, economic benefits and costs, rights, distributional equity and procedural fairness.

Bias and natural resource management

Like all people, environmental managers and natural resource managers are subject to bias. Sayed Iftekhar and David Pannell recently explored the influence of bias in natural resource management (NRM) and found that it's possible to improve our performance if we recognise these biases and work to reduce them (Iftekhar & Pannell, 2015).

The take home message from their study is that NRM agencies need to be aware of the influence of biases when management decisions are undertaken. There are many things they can do that will help limit the impact of bias.

First, agencies need to promote a culture of learning. It needs to be recognized that both successful and failed projects generate valuable information about the management interventions. This could be done by providing appropriate incentives (tangible and intangible) for the managers and decision makers to consider the full range of options before making any decision, or asking managers to justify their decisions to external parties.

Second, adoption of a decision support system could facilitate retention and storing of relevant information. It may also make learning from past projects easier and help in systematic evidence-based decision making. Of course, relevant staff should be adequately trained and properly incentivized to use such systems.

Third, conducting benefit-cost analyses of planned options would help to refine and prioritize the options during the design phase of an adaptive management cycle. Benefit-cost analysis provides a systematic framework to include all relevant costs and benefits (both market and nonmarket goods and services) related to a project.

Fourth, involvement of external third-party reviewers may also help in designing more realistic and feasible projects.

And, **finally**, scenario analysis should be conducted as part of the assessment and design phase to anticipate the expected outcomes of different decision options.

It is advisable to consider the likely impacts of different types of biases, and the effectiveness of potential remedial measures before making any final recommendation for use in decision making for natural resources.



Sayed Iftekhar, on the right, listens to ecologist Geoff Kay in a grassy woodland (a threatened ecosystem). Sayed has investigated how NRM managers are often influenced by unacknowledged biases in their decision making. (Photo by David Salt)

Hidden agendas: There may be political or bureaucratic objectives unrelated to the public interest, so that research that seeks to advance the public interest is not given the priority it deserves.

Policy fashions and crises: Policy attention tends to be directed to certain issues with high currency, often issues where there is a perceived crisis, and this may leave little scope for research (or any other input) to influence policy in an area that is not currently high on the agenda.

On the other hand, in certain circumstances, it may be easier to foster policy change in an area that is not in the public spotlight, at least from the perspective of avoiding controversy. This depends on convincing policy makers that a change is justified, which is likely to be difficult for non-topical issues.

Difficulty getting access: Policy makers often prefer local trusted sources for information. This is understandable, given the flood of information that policy makers can face on some issues, but it does not ensure that the most appropriate information is used.

Distrust: There may be suspicion about motivations of researchers, so that they are treated as just another interest or lobby group. Even without overt advocacy, some environmental scientists tend to intertwine facts and values, and this affects the perceived independence of their scientific advice.

Lack of expertise: In some policy agencies there is rapid turnover or movement of staff leading to a lack of expertise in responsible staff and a lack of knowledge of relevant research and researchers. A culture or belief may develop in government agencies that detailed subject knowledge is not necessary.

Another set of challenges arises from differences between the circumstances, cultures and reward systems of researchers and policy makers.

Different outcomes sought: Researchers place a high value on knowledge and innovation while policy officers seek to advance the public interest, to capture resources, and to please their political masters, who are primarily concerned about being re-elected.

Source of recognition: The sources of recognition for practitioners are different: from an administrative or political master in the case of policy, and from peers in the case of research.

Achievements rewarded: Policy officers tend to be promoted based on their ability to successfully implement desired policy programs, while researchers are promoted according to their productivity of research outputs, especially of those judged to be high in scientific quality.

Controversy vs compromise: Policy officers aim to resolve management or political problems with minimal controversy, making pragmatic compromises wherever necessary, whereas a healthy scientific discipline thrives on open debate, and will not compromise the truth.

Communication: Researchers and policy makers speak different languages, with different acronyms and jargon and different hidden assumptions. Scientific communication can be hard to understand even between different scientific disciplines. Policy officers deal mainly with very brief, simply written and highly interpreted/synthesised material conveying only essentials with a focus on practical implications and recommendations. On the other hand, even brief scientific writing is considerably more detailed and qualified and much research deals with practical implications as an afterthought, if at all.

Time frame: Once the decision has been made to develop or change a policy, the time frame for the process is usually short. This can place participants under great pressure, with little time

To inform policy, speak to policy makers

How can researchers and policy-makers work together more effectively to narrow the gap between science and policy? Influencing policy is not a process as easily definable as, say, publishing a scientific paper. Policy outputs appear in a number of forms over a variety of timeframes, and are rarely tracked back to single meetings or workshops. The very process of influencing policy makers is difficult to define. Anyone who thinks it's a rational, linear process probably hasn't tried it.

About a decade ago a group of decision scientists from the Applied Environmental Decisions Analysis Hub (the precursor of CEED) ran a workshop with policy makers from the Department of the Environment to discuss: How can we communicate research discoveries to policy makers and managers at minimal cost? How can we find out what research questions may deliver answers needed by policy makers and managers? The answers emerging from that discussion can be read in Gibbons et al, 2008 or read about the discussion in [Decision Point #25](#), p8.

It was agreed that personal relationships and networks were key to effectively influencing the development of policy. Activities that would serve to help foster effective relationships and networks include the creation of policy buddies (ie, researchers nominating policy people they need to interact with on specific topics), having research staff sit in government departments and vice-versa, reviewing rewards to researchers for making the

extra effort to influence policy (currently there are few), creating mechanisms by which policy makers can alert researchers to their specific concerns, and contact mapping (ie, figuring out just who is in whose network).



A tale of two cultures. Researchers and policy-makers have different reward systems that should be considered when seeking engagement from one another.

for careful consideration or collection of information. Research is generally slow and unresponsive to urgent policy needs, although it can be responsive in the longer term.

Supply vs demand: Policy usually addresses a problem identified by someone else (demand-driven), while the directions of research are often selected by researchers (supply-driven), particularly in a university environment. In a government research agency, research directions may be a mix of supply- and demand-driven.

Complexity vs simplicity: Policy officers prefer simple, straightforward advice with few, if any, caveats, whereas researchers tend to enjoy unravelling the full complexity of an issue, with all caveats highlighted.

Specialisation vs breadth: University training and the academic reward system encourages narrow specialisation, whereas policy officers need to consider a broad range of factors.

People focus: Policy involves intensive interaction among diverse groups of people, requiring highly developed social skills, while for some scientists working intensively with people is not comfortable. Of course, good interpersonal skills are valuable in both realms, but I believe that there is a difference in the degree of their importance.

Continued on page 50



Go east young man! David Pannell (pictured above) is a Chief Investigator with CEED. He's a resource economist based at UWA who has put considerable time and effort into influencing environmental policy.

Decision scientists in the policy arena

The following strategies, adapted from Pannell and Roberts (2009) and Gibbons et al (2008), are provided as food for thought for decision scientists. They are not in order of priority and they shouldn't be considered as a prescriptive recipe. Rather, think of them more as ingredients. It depends on how you combine them as to what type of cake you create.

1. Develop relationships with policy makers: Attempt to establish a high level of mutual understanding and trust. Information needs to flow in both directions. Only that way can you understand their perspectives and needs, and they understand your contribution.

2. Research is not enough: Appreciate that good science is needed but is not sufficient for decision makers. In considering policy options, policy makers will probably be more concerned with social, economic, political or administrative aspects than with technical aspects of science.

3. Practice excellent communication: In communications, recognise the lack of time that policy makers have. Be very brief, focus on clear messages, use simple language, free of jargon, using a mixture of approaches. Written material is useful but is not sufficient. Even more important is effective verbal communication.

4. Simplicity is essential: As far as possible, the solutions one offers need to be simple, transparent and understandable. Policy makers are likely to be suspicious of solutions that rely on complex and opaque computer models (although the success of climate modellers in influencing the debate about climate change shows that models can be used if their message is sufficiently clear and simple).

5. Work with intended users: This will help to ensure that the solution being proposed is in fact practical and sufficiently simple. It will help to make sure that their issue of concern is addressed in a way that is relevant to them. When attempting to convince policy makers, it helps to be able to demonstrate that the solutions being proposed are already in use in the real world.

6. Distinguish between knowledge and values: Be clear that the values that policy attempts to enhance are based on the desires of the community, not those of researchers. It is acceptable for research to deal with values (eg, studies of the non-market values of environmental outcomes) but one must be clear that policy makers will have their own views about the values. Traditionally, science deals primarily with knowledge rather than values, but of course scientists are influenced by their own values.

7. Be pragmatic: One has to accept compromise, and it may be necessary to make conscious decisions about where you can and cannot afford to give ground.

8. Be patient and persistent: Your work may not be influential at first, but its acceptance could grow over time if you are persistent. Establish networks and build support for your ideas over time. Repetition is essential, even to people who are already on your side.

9. Be resilient: Numerous problems, frustration and setbacks will arise. People with vested interests in the status quo will actively resist proposals for change. These people may be insiders to the policy organisation and so have better access to decision makers than outside researchers do.

10. Timeliness is important: Be prepared to respond quickly to requests for information. Policy makers cannot wait for additional research.

11. Find a champion: Elsewhere I've noted mixed evidence on this, but at least in some cases it is likely to be worth cultivating a champion for your work within the policy organisation.

12. Avoid any appearance of vested interest: Do not present findings and seek funds at the same time.

13. Choose your research/analysis topics well: It doesn't matter how good your communication is or how strong your policy networks are if the topics you are researching or analysing are not important to policy makers, or are not providing information that can help them. This seems obvious, but is sometimes not considered sufficiently when decision scientists are planning their own work.

Given these many challenges, it might seem remarkable that research ever plays a meaningful role in the development of policy. Often it doesn't, but sometimes it does. Consider the following case studies.

Case studies of impact

Fiona Gibson (working with a range of colleagues, including me) recently analysed the factors that facilitated or inhibited the use of rigorous decision-support tools in Australian environmental policy and management (Gibson et al, 2017). We selected seven case studies of policy development and conducted personal interviews with managers and policy makers who had been directly involved.

The case studies we reviewed involved:

The Southern and Eastern Scalefish and Shark Fishery policy (perceived by interviewees to be an example of where decision-support tools were much used = high);

The Representative Areas Program (high);

The South West Marine Reserve Network (low);

The National Reserve System (none); and

Three examples of *Threatened Species Protection* in different jurisdictions, Australian national (none), New Zealand (moderate) and New South Wales (moderate).

The selection of case studies was not intended to be representative of all possible conservation policies. However,

they were a diverse selection and provided insights that are likely to be transferable to other case studies and policies.

Drawing from the existing literature, the researchers identified a range of factors that are likely to promote or prevent the uptake of decision support tools in environmental and conservation programs. I'd suggest that these can be interpreted as factors influencing the uptake of decision science more generally, not just decision-support tools.

- Presence of a champion for the tool within the agency
- Presence of an advocate for the tool outside of the agency
- Existence of a relationship between agency staff and tool experts
- Presence of large numbers of stakeholder groups affected by the policy outcome
- Ability of the tool to deal with missing information
- Whether the tool can be applied quickly
- Whether the policy process allows adequate time for tool use
- Whether the tool capabilities align with policy objectives and policy operation

These factors were used to develop the questions used in the policy-maker interviews. Those factors rated as most important by the interviewed policy officers and managers were the alignment of a decision support tool with policy objectives and operation, and its ability to be useful even when there is missing data.

Measuring science performance and policy impact

How effective has CEED been in influencing policy? People close to CEED are well aware that CEED researchers have made many important contributions to environmental policy and management. However, measuring these impacts is notoriously difficult.

Even if environmental policy or management have changed, it can be very difficult to know the extent to which the change can be attributed to research. Policy development is a complex and messy process, with many players involved. And the eventual impacts of policy change on environmental outcomes are often uncertain, unclear and delayed. Maybe that is why quantitative analyses of the impact of research on environmental management, environmental policy and environmental outcomes are rare.

Working with a small team from UWA, we tried to capture CEED's impacts, as well as documenting its academic outputs, collaborations and citations (Thamo et al, 2018). We hope our approach might assist other environmental research networks and centres to measure the influence of their own research efforts.

The CEED impact evaluation collected data on 87 CEED projects and discussed nine of these in detail. It found that there was high academic performance in many of CEED's outputs and high policy and management impact in some projects, but not all.

A number of important lessons and implications were identified in the impact analysis.

There have been many studies on the factors that underpin research impact, and most of them highlight the importance of engagement and good relationships with research users, the quality of communication (eg, see [Decision Point #73](#) and [Decision Point #74](#)).

However, the evaluation found that just as important was what research is actually done. If research is not providing insight or tools that are actually useful to policy makers or managers, even strong relationships and excellent communications won't lead to impact.

Therefore, developing a research culture that values impact and considers how it may be achieved prior to the selection of research projects is potentially important. The role of the centre leadership team in this is critical. Embedding impact into the culture of a centre probably happens more effectively if expertise in research evaluation is available internally, either through training or appointments.

A challenge in conducting this analysis was obtaining information related to engagement and impact. There may be merits in institutionalising the collection of impact-related data from early in the life of a new research centre.

In this analysis, there was little correlation between academic performance and impact on policy and management. It should not be presumed that the most impactful projects will be those of greatest academic performance.

Finally, there are often long time lags between commencing research and delivering impact – decades in many cases. Therefore, there is a need to allow the longest possible time lag when assessing impact. On shorter timescales, it may be possible to detect engagement, but not the full impact that will eventually result.

The issue of alignment with objectives arose because in some cases a tool was perceived to be focused on environmental outcomes and not able to adequately represent non-environmental policy objectives.

Lack of complete data is almost always a challenge when considering policy options, and the tools that were viewed favourably and utilised more extensively tended to be viewed as being amenable to strategies such as expert elicitation to fill data gaps.

The least important factors, at least for these interviewees, were the presence of a champion of the decision-support tool within the management agency, and the time required to apply the tool. The finding regarding an internal champion conflicts directly with some previous studies where the presence of a champion was critical to effective use of the tool (eg, consider Richard Maloney's discussion on success factors behind the implementation of the Project Prioritisation Protocol in which he cites champions as being of critical importance, see [Decision Point #76](#)).

Interestingly, the interviews revealed additional important factors that had not previously been identified in the literature. These included: the existence of multiple (potentially unstated) policy objectives, and the autonomy of the agency. Where there were two conflicting objectives (eg, conservation and sustainable economic use of the resource), some interviewees lacked confidence that the tool could handle this, or would give the competing objectives weights consistent with those of the policy makers. It was felt by some interviewees that relatively autonomous government agencies are less prone to intervention by a government minister concerned with the politics of an issue. This lower risk of interference made it easier for transparent and systematic decision processes to operate. The particular autonomous bodies in these case studies also had a greater emphasis on day-to-day engagement with stakeholders, such that the potential benefits of a decision support tool in enhancing engagement may have been more apparent.

The findings of Gibson et al (2017) reflect the perceptions of the interviewees. Notably, it appeared that some of the most negative perceptions about particular tools may have arisen from misunderstandings about the tools, or a lack of capability to use the tool appropriately. There were examples where a tool that was rated negatively by one agency for a particular criterion (eg, ability to cope with missing data), was rated positively by another agency for the same criterion, and the other agency had actually used it successfully. This highlights the importance of clear communication channels and the provision of training and support to allow the decision tools to deliver their potential.

So, if you are a decision scientist, how might you make a bigger impact when it comes to the development of policy? In the box on 'decision scientists in the policy arena' I outline a set of strategies you might consider. It is not a prescriptive or complete list but these 13 strategies are important considerations for any researcher wanting to influence policy.

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References and recommended reading

- Gibbons P, C Zammit, K Youngentob, HP Possingham, DB Lindenmayer, S Bekessy, M Burgman, M Colyvan, M Considine, A Felton, RJ Hobbs, K Hurley, C McAlpine, MA McCarthy, J Moore, D Robinson, D Salt & B Wintle B (2008). Some practical suggestions for improving engagement between researchers and policy-makers in natural resource management. *Ecological Management and Restoration* 9: 182-186. For a discussion on this paper see [Decision Point #25](#), p8.
- Gibson FL, AA Rogers, ADM Smith, A Roberts, HP Possingham, M McCarthy & DJ Pannell (2017). Factors influencing the use of decision support tools in the development and design of conservation policy. *Environmental Science and Policy* 70: 1-8.
- Iftexhar MS & DJ Pannell (2015). 'Biases' in adaptive natural resource management. *Conservation Letters* 8: 388-396. For a discussion on this paper see [Decision Point #93](#)
- Kahneman D (2011). *Thinking, Fast and Slow*. Farrar, Straus and Giroux, New York.
- Maron M, JR Rhodes & P Gibbons (2013). Calculating the benefit of conservation actions. *Conservation Letters* 6: 359-367. For a discussion on this paper see [Decision Point #69](#), p10,11.
- May J, RJ Hobbs & LE Valentine (2017). Are offsets effective? An evaluation of recent environmental offsets in Western Australia. *Biological Conservation* 206: 249-257.
- McDonald JA, J Carwardine, LN Joseph, CJ Klein, TM Rout, JEM Watson, ST Garnett, MA McCarthy & HP Possingham (2015). Improving policy efficiency and effectiveness to save more species: A case study of the megadiverse country Australia. *Biological Conservation* 182: 102-108.
- Pannell DJ & AM Roberts (2009). Conducting and delivering integrated research to influence land-use policy: salinity policy in Australia. *Environmental Science and Policy* 12: 1088-1099.
- Pannell DJ & FL Gibson (2016). The environmental cost of using poor decision metrics to prioritise environmental projects. *Conservation Biology* 30: 382-391. For a discussion on this paper see [Decision Point #82](#)
- Schwartz MW, CN Cook, RL Pressey, AS Pullin, MC Runge, N Salafsky, WJ Sutherland & MA Williamson (2017). Decision support frameworks and tools for conservation. *Conservation Letters* 11 <https://doi.org/10.1111/conl.12385>
- Clark RN, EE Meidinger, G Miller, J Rayner, M Layseca, S Monreal, J Fernandez & MA Shannon (1998). Integrating science and policy in natural resource management: lessons and opportunities from North America. *Gen. Tech. Rep. PNW-GTR-441*. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.
- Mumford T & N Harvey (2014). Champions as influencers of science uptake into Australian coastal zone policy. *Coastal Management* 42: 495-511.
- Rose DC, N Mukherjee, BI Simmons, ER Tew, RJ Robertson, ABM Vadrot, R Doubleday & WJ Sutherland (2017). Policy windows for the environment: Tips for improving the uptake of scientific knowledge. *Environmental Science and Policy* <https://doi.org/10.1016/j.envsci.2017.07.013>
- Thamo T, T Harold, M Polyakov & D Pannell (2018). *Assessment of Engagement and Impact for the ARC Centre of Excellence for Environmental Decisions*. University of Western Australia.



Collaboration across the sea

Better marine conservation outcomes from a little cross-boundary collaboration

By Tessa Mazor (University of Queensland), Noam Levin (Hebrew University of Jerusalem) and Salit Kark (University of Queensland)

Conservation decisions are usually made at the national scale, but biodiversity doesn't stop at national boundaries – and neither do conservation threats and opportunities. Cross-boundary conservation issues are particularly relevant in marine environments. Marine boundaries are diverse, and range from coastal to territorial waters to Exclusive Economic Zones and the High Seas. And they all have different roles in shaping marine conservation outcomes.

Above: Gazing across the Mediterranean Sea, home to hundreds of species of threatened species and regarded as a biodiversity hotspot. Protecting this natural value is beyond the scope of any single nation lining its shores. How do you approach the task of collaboration in a such a complex sociopolitical environment?

Many activities that impact biodiversity – such as fisheries, pollution and oil mining – cross borders. Thus, environmental and conservation decisions can be substantially influenced by whether they cross international and political boundaries.

Work over recent years is helping us better understand how collaboration can shape conservation across marine boundaries. One of the most important inclusions into marine conservation has been the dynamics of marine uses and socio-economic activities; these have been shown to sometimes be central to conservation outcomes in marine environments. In this article we will review recent advances, open questions and directions in the area of marine conservation collaboration, and provide case studies from CEED-led work over recent years.

Types of conservation collaboration

Collaboration in conservation decision making is seen as a key to success in many areas and realms (see the box on 'collaboration is an international goal'). But how is the scientific community responding? And what tools do we have on hand?

Recent research around the globe aims to improve our understanding of the value of collaborations for conservation. Salit Kark and colleagues (2015a) reviewed the numerous circumstances in which conservation collaboration is applicable across terrestrial and marine realms. Collaboration can occur at various spatial scales and can be diverse, ranging

Key messages:

1. Decisions around marine conservation can be substantially influenced by whether they cross international and political boundaries.
2. Taking this into account, it is important to understand the challenges, opportunities and constraints involved in conservation collaborations.
3. Collaboration can occur at a variety of spatial scales (regions, states, countries), and involve a range of different organisations and stakeholders. Different planning approaches and tools apply in different situations.
4. Systematic conservation planning can provide a valuable framework for deciding priorities in these multi-country regions but needs to take into account the complexity of multiple jurisdictions.
5. Important advances have been made in the area of marine conservation collaboration in recent years.

Collaboration is an international goal

Collaboration lies at the heart of some of the world's most important international conventions on sustainability and conservation. The United Nation's Sustainable Development Goal 17 emphasises partnerships and collaboration to achieve the sustainable development agenda. It calls upon multi-stakeholder partnerships to transfer knowledge and share resources (United Nations 2015). Likewise, 'Strategic Goal E' (Target 17-20) of the Convention on Biological Diversity's Aichi Biodiversity Targets addresses the need to enhance stakeholder's participation and increase the dispersion, sharing and application of science and technology (Convention on Biological Diversity 2010).

from collaboration across-borders (such as regions, states or countries), organisations (universities, NGOs, governments) or stakeholders (local community members, indigenous communities, government agencies, conservation groups, recreational groups, scientists).

Today, conservation organisations are more numerous than ever pursuing various objectives, often within the same space (Bode et al, 2010; Gordon et al, 2012). So too is the growth in initiatives to implement transboundary parks and protected area networks that span multiple countries (Chester 2012). Thus with increasing conservation efforts, combined with limited funds and restricted budgets, we must reap the most from collaborations and ensure they are effective as possible.

Collaboration in marine systems

The issue of collaboration is particularly relevant for marine systems. Marine ecosystems are inter-connected through ocean currents, circulations and larval dispersal (Trembl & Halpin, 2012), with borders less defined in comparison to terrestrial systems. Many countries have not declared their boundaries (Exclusive Economic Zones (EEZ)), and there are the 'high seas' which is ocean that does not fall under any countries' jurisdiction.

Geomorphology of marine environments such as enclosed or semi-enclosed seas (eg, The Mediterranean Sea, The Baltic Sea, The North Sea), gulfs and basins (eg, Gulf of Mexico, Gulf of Alaska) where numerous states or countries rely on the same shared resource automatically means collaborations are essential. For example, in the North Sea coordinated efforts between countries were taken to jointly survey macrozoobenthos communities to monitor habitat changes and impacts from climate change and fishing ([North Sea Benthos Project](#)), as well as to monitor oil pollution using aerial surveys (Carpenter, 2007).

Similarly, marine species have no borders. Countries share species as they move and migrate across waters. Iconic species that migrate across multiple countries include sea turtles, tuna, dolphins, shorebirds and whales (eg, Mazar et al, 2016). The survival of such species often relies upon their ability to migrate across the ocean to breed or feed. Fish stocks are also often shared across countries (Kroodsma et al, 2018) (eg, Northeast Arctic cod stock shared between Russia and Norway), and spawning grounds may be located in areas of different jurisdictions. Therefore, coordinated actions are critical to conserve or protect marine species.

The high connectivity of marine systems also means that marine

threats span multiple countries. For example, an oil spill in one country can impact another (Goldman et al, 2015). One of the most infamous examples of this is the Deep Horizon Oil spill in the Gulf of Mexico where impacts were spread across four states; Louisiana, Alabama, Mississippi and Florida. Fishing boats aren't easily tracked and illegal fishing and overexploitation of fishery stocks is difficult to control. Pollution (eg, plastics and waste) and the release of toxic substances in waters of one country can flow to that of another.

Given the connectivity, species and threats shared across the marine realm, tailored approaches and tools need to be developed and shared in order to protect and conserve marine biodiversity and to ensure efforts taken in one place are not being hindered in another.

Tools for marine conservation collaboration

Here, we review recent advancements over the past five years in the area of cross boundary collaboration in marine conservation (with a focus on the contributions of the ARC Centre of Excellence for Environmental Decisions (CEED)). These initiatives improve the efficiency and effectiveness of marine conservation collaboration and decision making in this vast and often understudied and unprotected realm.

The Mediterranean Marine Conservation Planning (MMCP) Initiative

The Mediterranean Sea is a biodiversity hotspot. It is surrounded by over 20 countries with a diversity of languages, religions, politics and economies. It is a prime example of a region requiring collaborative efforts, and there have been various attempts to forge collaborations. One example is MedPAN, an NGO that promotes coordinated efforts to establish a network of marine protected areas in the Mediterranean.

Yet, systematic conservation planning in the region has lagged behind, with the first localised efforts less than 10 years ago (eg, Giakoumi et al, 2011). The complex setting of the Mediterranean coupled with its lack of, and need for, conservation planning, has prompted the Mediterranean Marine Conservation Planning (MMCP) initiative. This initiative was established by our team and has been the focus of a number of international workshops and publications.

We started the MMCP initiative in 2012. It established a group of scientists and practitioners from across the Mediterranean Sea working on conservation planning, to help advance cross-boundary collaboration and spatial planning in this biodiversity-rich and highly threatened hotspot. The first workshop was held in 2012 in Santorini (Giakoumi et al, 2012), which was followed by two additional workshops (in Nahsholim, Israel in 2013; and in Lecce, Italy in 2015).

This collaborative work has since developed and enhanced the establishment of the more formal MarCons COST action (<http://www.marcons-cost.eu/>) under the European Union Horizon 2020 framework programme.

The 'Mediterranean melting pot'

The Mediterranean Sea supports a rich biodiversity (Fig 1) but faces many threats that have to be dealt with by multiple nations. The Sea is surrounded by over twenty countries spread across three continents. It is visited by around 200 million tourists a year and supports the livelihood of some 150 million people via small-scale subsistence fishing, employment within commercial fisheries and as a food source. In addition, the multiple users of this common resource face very different circumstances. Countries surrounding the Mediterranean Sea show a vast array of cultural values, economic statuses, political systems, religions and languages. All these additional factors can impede successful collaboration (see the box on collaborative potential).

Most conservation efforts in the Mediterranean Sea are uncoordinated and are not protecting the sea's highly threatened biodiversity. With limited conservation measures in place, the sea's native species and ecosystems continue to face threats from both land- and sea-based human activities. Existing marine protected areas (MPAs) are relatively small and are not based on coordinated legislation or criteria for establishment; each country has its own guidelines for administering MPAs. While the implementation of protected areas has raised conservation awareness, limited structural integrity and cross-country collaboration challenge the ability of these areas to protect and sustain the biodiversity of the Mediterranean Sea.

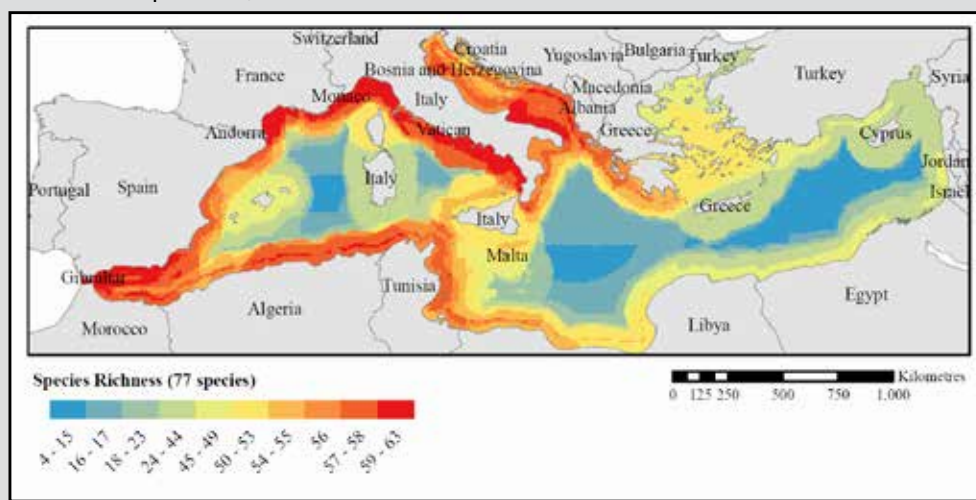


Figure 1: Species richness of 77 threatened vertebrates in the Mediterranean Sea. If the Sea was under the control of a single country, planning an effective reserve system to protect these threatened species would be considerably simpler than working with the actual situation of 20 countries, each with different rights, values and histories in relation to the Mediterranean. (From Mazor et al, 2013)

Within the context of the Mediterranean, we outline and summarise work undertaken by CEED, focusing on the approaches and tools for conducting collaborative conservation.

Frameworks: Structured frameworks presented in Kark et al, (2015a; and see Figure 1) and in Micheli et al, (2013; see Figure 7) aim to improve the incorporation of collaboration into conservation and planning. Below we follow the general outline proposed within these frameworks.

Spatial Data Collation: Given the complexity of the region and its many countries, spatial marine information rarely covers the entire sea, thus limiting the extent of broad scale collaborative projects. Noam Levin and colleagues (2014) outlined the status of knowledge for three types of spatial information; bathymetry, classification of marine habitats, and species distributions. Findings of this study suggest data are most readily available and of sufficient quality across the western European countries, and calls for more collaborative networks and databases such as EurOBIS to facilitate collaborations.

Threat Assessments: Often conservation collaboration is necessary to prevent a threatening activity. For example, the increase in oil and gas extraction within the Mediterranean region has enhanced collaborations between countries (Kark et al, 2015b; Mazor et al, 2018). New natural gas discoveries in the eastern Mediterranean have led to an agreement between Cyprus and Israel in determining their maritime boundaries.

Similarly, the shared threat of biological invasions requires coordinated efforts. The Mediterranean Sea has approximately

1000 alien species and numbers are expected to grow with the enlargement of the Suez Canal (Giakoumi et al, 2016). Given the fluidity and intrinsic connectivity of the marine systems means managing threats require novel ways to address them such as three-dimensional planning (Levin et al, 2018). Prioritising conservation areas in 3D enables threats and multi-users of the ocean space to be explicitly targeted at different depths (Venegas-Li et al, 2017).

Collaboration Potential: Before collaborative projects are undertaken or formalised, Levin and colleagues (2013) presents a tool to estimate the potential for collaboration success (see the box on collaborative potential). This method incorporates understanding about current and historical political, trade and signed conventions and links between given countries. This allows an evaluation to help determine which partnerships will be most viable for conservation collaboration success.

Costs and Benefits: Once collaboration partners have been established it is important to determine the costs and benefits of such partnerships. In the Mediterranean Sea where economies are vastly different and there is a heterogeneous spread of species, conservation costs will not be equal. Tessa Mazor and colleagues (2013) highlighted the huge monetary gains by coordinated collaboration across the Mediterranean Sea, where over two thirds of the cost can be saved by a collaborative plan compared to the costs were countries to act individually.

However, benefits and costs will not be the same for all countries, some will have to contribute more and receive less

direct conservation benefit within their own waters, whereas others will reap large conservation benefits for less cost. Hence, such assessments can be helpful for prior assessments on the feasibility of conservation collaborations and for understanding the required inputs from each collaborator. Similarly, Gissi and colleagues, (2018) applied systematic conservation planning in the Adriatic-Ionian Region and used the Protection Equality metric (Chauvenet et al, 2017) to understand how equally countries and industries are protected. The study found that countries maintained a high level of equity but it was lower for industries.

Stakeholder Collaborations: The large number of protected area proposals for the region unfortunately has caused more ambiguity and confusion than progress. To resurrect the multitude of overlapping agendas, Micheli and colleagues (2013) synthesised for the first time 12 large-scale plans across the region, dissolving these into one plan of overlapping priority conservation areas for the Mediterranean Sea. The resulting core areas can help identify major regions of value for multiple initiatives and can serve as a way to incite collaboration across all partners – being win-win for all.

Besides organisational stakeholders, there are also stakeholders of the marine environment, which constitute another mass of actors and members. The ocean is often viewed as a vast open space, but exploring the territorial waters of a country and the number of users and activities quickly dispels this vision. The crowded sea of Israel's Mediterranean Sea was a case study used by Mazar et al (2014) to illustrate exactly how busy the marine waters can get. This work presents the tool Marxan with Zones

as an approach to explore and quantify the trade-offs between different stakeholder objectives such as species protection, hydrocarbon extraction and fishery revenue. While applied at a country scale, this approach can be expanded to wider regions and areas, and helps to understand the ability for stakeholders to collaborate and the trade-offs that they may need to take.

Given the new rise in marine spatial planning programs across the globe, this pre-analysis of trade-off scenarios can be very helpful in ensuring implementation success, where trade-offs are made explicit and transparent early on in the process.

Transboundary Marine Protected Areas

The establishment of protected areas are the primary tool to halt biodiversity loss. Thus, the most common form of cross-boundary collaboration is the establishment of transboundary protected area or networks. Although these areas sounds like a grand solution to the biodiversity crisis, implementing such areas are not straight forward (Guerrero et al, 2010). The complexities often lie in the economic and socio-political differences between countries. For some parts of the world, branding transboundary parks as 'peace parks' may be one solution for gaining political interest (eg, The Red Sea Marine Peace Park between Israel and Jordan, Spratly Islands Marine Peace Park in the South China Sea) but moving beyond a paper parks presents more challenges. Several CEED-related projects involving investigations from various parts of the globe have helped inform how to overcome hurdles relating to the establishment of transboundary parks.

Collaborative potential

International collaboration has been shown to be a key to success in tackling a range of environmental problems. Furthermore, it has been found that collaboration in conservation has the potential to substantially reduce costs. While, these factors may be enough to encourage some countries to engage in conservation, for others there are large cultural differences, political histories and language barriers that may be too difficult to overcome. For establishing collaborative conservation initiatives where do we begin? How can we identify countries which have the potential to collaborate successfully in conservation?

To address this, Noam Levin and colleagues have developed a framework for including collaborative potential into conservation planning (Levin et al, 2013). Because there is no way of calculating the success of future collaborations, they assessed current connections and linkages between countries and used these as surrogates. The surrogates they used included demographics, socioeconomic (eg, trade and tourism), political (eg, history of conflicts) and historical features.

Given the large heterogeneity among countries surrounding the Mediterranean Sea (and the fact that collaboration for marine conservation is necessary within this shared environment), it made a good case study for the method. However, the findings from this study could be applied to any area where collaboration for conservation is needed.

Priority areas for conservation may look promising on paper, but they may not be actually achievable. By incorporating collaborative potential the researchers found that the spatial



Figure 2: A map showing military conflicts and democratic tendencies in countries surrounding the Mediterranean Sea. These are just two of the socio-political dimensions being factored into the collaborative potential framework devised by Levin and colleagues (2013). (From Levin et al, 2013)

priorities for marine conservation in the Mediterranean Sea shifted to the northern part of the Mediterranean Sea, where collaboration between countries (and especially within the European Union) is well established.

This type of analysis allows planners and decision makers to incorporate feasibility when setting up marine or terrestrial trans-boundary park and international conservation projects. Besides this, it can help us realise which areas may need extra resources and time for facilitating collaborative conservation.

Collaboration and oil extraction

Oil and gas extraction in the marine realm is a good example of an activity that presents threats to biodiversity that transcend national boundaries. What is sometimes overlooked is that it also presents opportunities. To mitigate the threats and seize the opportunities requires effective collaboration.

Salit Kark and colleagues (2015b) reviewed the risks and impacts of offshore oil and gas extraction globally, and discussed how the conservation community can be better prepared. They also reflected on some of the conservation challenges and opportunities arising from offshore hydrocarbon development. These challenges include threats to ecosystems and marine species from exploration, oil spills, and operations infrastructure (in both marine and coastal areas). They discussed impacts on native biodiversity from invasive species colonising drilling infrastructure, and increased political conflicts that can delay conservation actions.

However, it's not all 'downside'. The expansion of offshore operations also brings with it potential opportunities that might be leveraged for conservation. Options include the use of facilities and costly equipment of the deep and ultra-deep hydrocarbon industry for deep-sea conservation research and monitoring, and the establishment of new conservation research, practice, and monitoring funds and environmental offsetting schemes. Collaborations have already begun in some regions, and in some cases involves global and local NGOs and other stakeholders.



The Tamar gas field production platform in the Mediterranean Sea. Oil and gas extraction in the marine realm is a good example of an activity that presents threats to biodiversity that transcend national boundaries. To mitigate the threats and seize the opportunities requires effective collaboration.

Pascual and colleagues (2016) present a study that surveys stakeholders across the Mediterranean Sea and Black Sea to determine perceptions of socioeconomic impacts on MPAs. Incorporating the perceptions of multiple stakeholders in the design of MPAs means more feasible and socially accepted conservation outcomes. This process can help find commonalities as well as identify areas of disagreement or possible conflict among stakeholders. Similarly, Beger and colleagues (2015) uses a multi-objective approach, setting six conservation objectives for creating a network of cohesive marine-protected areas in the Coral Triangle (see [Decision Point #96](#)).

Conflict can hamper the potential for transboundary conservation collaborations. Levin and colleagues (2018) explored relationships between countries in the Western Indian Ocean to help aid the establishment of transboundary marine protected areas such as the Lubombo Ponta do Ouro-Kosi Bay Marine and Coastal Transfrontier Conservation and Resource Area, established in 2009 and the future Ruvuma-Palma National Reserve. The likelihood of poor collaborations was inferred from compiling data from historical conflicts and wars in the region and anti-shipping activities (piracy), whereas positive connections were inferred from trade-agreements as well as multi-lateral and bilateral maritime and conservation agreements. Understanding ecological versus institutional linkages between countries has also been explored in the Indo-West Pacific (Tremel et al, 2015). Using a quantitative network-analysis method, the approach helps to reveal areas of misalignment where strengthening institutional ties can help improve important ecological processes.

Collaboration for better conservation outcomes

Addressing large-scale biodiversity loss requires large-scale actions. Conservation collaboration is one important ingredient in achieving this.

In this article we have outlined a number of useful tools and approaches for evaluating efforts aimed at implementing collaborative marine conservation (summarised in Table 1).

Our work within the Mediterranean Sea has been successful in advancing conservation planning and collaboration in this region. We have demonstrated the value of these approaches. However, these applications need to be refined and adapted to other parts of the globe.

Given that collaborations can be hugely beneficial for conservation, we should make every effort to ensure the process is fruitful, timely and cost effective.

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Reference

- Beger M, J McGowan, EA Trembl, AL Green, AT White, NH Wolff, CJ Klein, PJ Mumby & HP Possingham (2015). Integrating regional conservation priorities for multiple objectives into national policy. *Nature communications* 6: 8208.
- Bode M, W Probert, WR Turner, KA Wilson & O Venter (2011) Conservation planning with multiple organizations and objectives. *Conservation Biology* 25: 295-304.
- Carpenter A (2007). The Bonn agreement aerial surveillance programme: trends in North Sea oil pollution 1986–2004. *Marine pollution bulletin* 54: 149-163.
- Convention on Biological Diversity (2010) Decision X/2, *The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets*.

Collaboration Type	Across Borders	Organisations	Stakeholders
Approaches and Tools	Collaboration potential network Transboundary parks Systematic conservation planning scenarios of costs and benefits	Game theory and simulation models Conservation planning tools Overlap analysis	Quantifying trade-offs between stakeholders (Marxan with Zones) Protection Equality Theoretical networks 3D Prioritisation
Key Literature	Levin et al. 2013 Levin et al. 2018 Mazor et al. 2013 Trembl et al. 2015 Beger et al. 2015 Kark et al. 2015a	Bode et al. 2010 Gordon et al. 2013 Micheli et al. 2013	Mazor et al. 2014 Klein et al. 2009 Gissi et al. 2018 Guerrero et al. 2014 Venegas-Li et al. 2017

Table 1: Approaches and tools for addressing collaboration in marine conservation. Note some of these are applied in terrestrial settings but can still apply to a marine context.

Chauvenet AL, CD Kuempel, J McGowan, M Beger & HP Possingham (2017). Methods for Calculating Protection Equality for conservation planning. *PLoS One* e0171591

Chester CC (2012). *Conservation across borders: biodiversity in an interdependent world*. Island Press.

Giakoumi S, HS Grantham, GD Kokkoris & HP Possingham (2011). Designing a network of marine reserves in the Mediterranean Sea with limited socio-economic data. *Biological Conservation* 144: 753-763.

Giakoumi S, T Mazor, S Fraschetti, S Kark, M Portman, M Coll, J Steenbeek & HP Possingham (2012). Advancing marine conservation planning in the Mediterranean Sea. *Reviews in Fish Biology and Fisheries* 22: 943-949.

Giakoumi S, F Guilhaumon, S Kark, A Terlizzi, J Claudet, S Felline, C Cerrano, M Coll, R Danovaro, S Fraschetti & D Koutsoubas (2016). Space invaders; biological invasions in marine conservation planning. *Divers & Distributions* 22: 1220-1231.

Gissi E, J McGowan, C Venier et al (2018). Addressing transboundary conservation challenges through marine spatial prioritization. *Conservation Biology* 32: 1107-1117.

Goldman R, E Biton, E Brokovich, S Kark & N Levin (2015). Oil spill contamination probability in the southeastern Levantine basin. *Marine pollution bulletin* 91: 347-356.

Gordon A, L Bastin, WT Langford, AM Lechner & SA Bekessy (2013). Simulating the value of collaboration in multi-actor conservation planning. *Ecological modelling* 249: 19-25.

Guerreiro J, A Chircop, C Grilo, A Viras, R Ribeiro & R van der Elst (2010). Establishing a transboundary network of marine protected areas: Diplomatic and management options for the east African context. *Marine Policy* 34: 896-910.

Kark S, A Tulloch, A Gordon, T Mazor, N Bunnefeld & N Levin (2015a). Cross-boundary collaboration: key to the conservation puzzle. *Current Opinion in Environmental Sustainability* 12: 12-24.

Kark S, E Brokovich, T Mazor & N Levin (2015b). Emerging conservation challenges and prospects in an era of offshore hydrocarbon exploration and exploitation. *Conservation biology* 29: 1573-1585.

Kroodsma DA, J Mayorga, T Hochberg, NA Miller, K Boerder, F Ferretti, ... & P Woods (2018). Tracking the global footprint of fisheries. *Science* 359: 904-908.

Levin N, AI Tulloch, A Gordon, T Mazor, N Bunnefeld & S Kark (2013). Incorporating socioeconomic and political drivers of international collaboration into marine conservation planning. *BioScience* 63: 547-563.

Levin N, M Coll, S Fraschetti, G Gal et al (2014). Biodiversity data requirements for systematic conservation planning in the Mediterranean Sea. *Mar Ecol Prog Ser* 508: 261-281.

Levin N, S Kark & R Danovaro (2017). Adding the Third Dimension to Marine Conservation. *Conservation Letters* 11: e12408

Levin N, M Beger, J Maina, T McClanahan & S Kark (2018). Evaluating the potential for transboundary management of marine biodiversity in the Western Indian Ocean. *Australasian Journal of Environmental Management* 25: 62-85.

Mazor T, HP Possingham, D Edelist, E Brokovich & Salit Kark (2014). The Crowded Sea: Incorporating Multiple Marine Activities in Conservation Plans Can Significantly Alter Spatial Priorities. *PLOS One* <https://doi.org/10.1371/journal.pone.0104489>

Mazor T, M Beger, J McGowan, HP Possingham & S Kark (2016). The value of migration information for conservation prioritization of sea turtles in the Mediterranean. *Global Ecology and Biogeography* 25: 540-552.

Mazor T, N Levin, E Brokovich & S Kark (2018). Conservation challenges in the face of new hydrocarbon discoveries in the Mediterranean Sea. *Offshore Ener & Mar Spat Plan* 260: 260-273.

Micheli F, N Levin, S Giakoumi, S Katsanevakis, A Abdulla, M Coll, S Fraschetti, S Kark, D Koutsoubas, P Mackelworth & L Maiorano (2013). Setting priorities for regional conservation planning in the Mediterranean Sea. *PLoS One* 8: p.e59038.

Pascual M, M Rossetto, E Ojea et al (2016). Socioeconomic impacts of marine protected areas in the Mediterranean and Black Seas. *Ocean & Coastal Management* 133: 1-10.

Trembl EA & PN Halpin (2012). Marine population connectivity identifies ecological neighbors for conservation planning in the Coral Triangle. *Conservation Letters* 5: 441-449.

Trembl EA, PI Fidelman, S Kininmonth, JA Ekstrom & Ö Bodin (2015). Analyzing the (mis) fit between the institutional and ecological networks of the Indo-West Pacific. *Global Environmental Change* 31: 263-271.

Venegas-Li R, N Levin, HP Possingham & S Kark (2018). 3D spatial conservation prioritisation: Accounting for depth in marine environments. *Methods in Ecology and Evolution* 9: 773-784.



Human and social dimensions of good environmental decision making

Socio-ecological analysis and modelling for better environmental outcomes

By Sarah Bekessy (RMIT University)

Since the management of the environment is, by definition, a social and political process, it stands to reason that responses to environmental problems must focus at least in part on questions of human behaviour. Environmental managers need to know how their interventions alter the motivations and thus the behaviour of people causing environmental problems. Yet social processes are often neglected by scientists and in real-world environmental decision-making. This neglect compromises the effectiveness and efficiency of the global investment in environmental interventions by governments, businesses and individuals.

Key messages:

1. Spatial conservation planning can be improved by incorporating multiple objectives, multiple actors and multiple uncertainties.
2. Incorporating human and social dimensions into the evaluation of environmental policy options increases the robustness of policy decisions.
3. Engagement with psychologists and social scientists can improve our ability to anticipate responses to policies, including feedback loops and perverse outcomes.
4. We are only just getting started in applying socio-ecological thinking to environmental decision making. Many innovations and new opportunities are emerging as new disciplines like behavioural economics are applied.

Above: A scene common in many Australian peri-urban zones—new houses, open fields, scattered trees and remnant patches of native vegetation. This landscape seems unremarkable but is actually a complex mosaic of biological, legal, economic and social values. Developing the economic potential of these landscapes doesn't have to occur at the expense of its biological values. (Photo by Phil Gibbons)

The challenges of holistically managing social-ecological systems are typically complex, dynamic and multifaceted. Therefore, conceptual and quantitative models are important for understanding and characterising socio-ecological systems, and for making predictions about the outcomes of management. The Socio-Ecological Analysis and Modelling Theme of CEED builds on techniques from a range of disciplines to analyse, model and integrate knowledge about socio-economic and ecological processes to improve environmental decision-making.

Here we present some of the highlights of this research theme and describe the evolution in thinking and method development that has occurred over the seven years of the Centre's life.

Considering multiple objectives, actors and uncertainties in planning

Spatial conservation planning can be improved by considering multiple objectives, multiple actors and multiple uncertainties. In so doing, conservation planning is simply aligning with real world practice.

Conservation planners need tools that match the world they are trying to plan for. This includes real-world complexities, constraints, uncertainties and political realities. Many

conservation initiatives do not fit the single-actor, single-action model underpinning most conservation planning tools. This is particularly a problem in complex planning environments, such as the urban-fringe (consider the planning work done in Bekessy et al, 2012; see the box on Planning for biodiversity and development in the urban fringe) or highly modified agricultural landscapes, where high biodiversity values must compete with multiple social and economic interests.

Over the lifetime of CEED, Centre researchers have addressed important gaps between conservation theory and real-world practice by considering multiple actors, multiple conservation actions and multiple uncertainties. Some examples of this work include our research on collaboration and the incorporation of social values into conservation planning.

Socially-acceptable conservation planning

Understanding how society perceives and values different areas of the landscape is important for effective land-use planning. Indeed, making use of social values is arguably one of the most important challenges in modern conservation planning, yet their potential remains poorly exploited. Previous research suggests the inclusion of social values can reduce conflicts between stakeholders and enable a more efficient implementation of conservation actions. However, the potential trade-offs related to incorporating social values into spatial conservation planning are not well understood.

Amy Whitehead and colleagues collected spatial data on social values by conducting a Public Participation GIS (Geographical Information System) survey in the Lower Hunter Valley in eastern New South Wales (Whitehead et al, 2014). Local residents were asked to map areas perceived to be important for their natural or potential development values. Randomly-selected landowners were given a map of the region and a set of sticker dots that corresponded to different social values, including biodiversity, natural significance and intrinsic value types (social values for biodiversity). Another set of stickers corresponded to areas believed to be appropriate for different types of future development (social values for development). Participants were asked to place stickers on areas of the map

Optimising resource allocation through collaboration to achieve conservation objectives

In a setting where multiple actors have similar conservation objectives, strategic collaboration may optimise their resources to achieve conservation objectives, thus freeing up resources for additional work. Gordon et al (2013) demonstrate that the cost savings from collaboration could vary significantly in different situations, ranging from a given actor making almost no savings through to saving almost 40% of the cost of achieving their objectives in isolation. The largest potential gains from collaboration occur when the conservation objectives of the two actors involved non-overlapping sets of species.

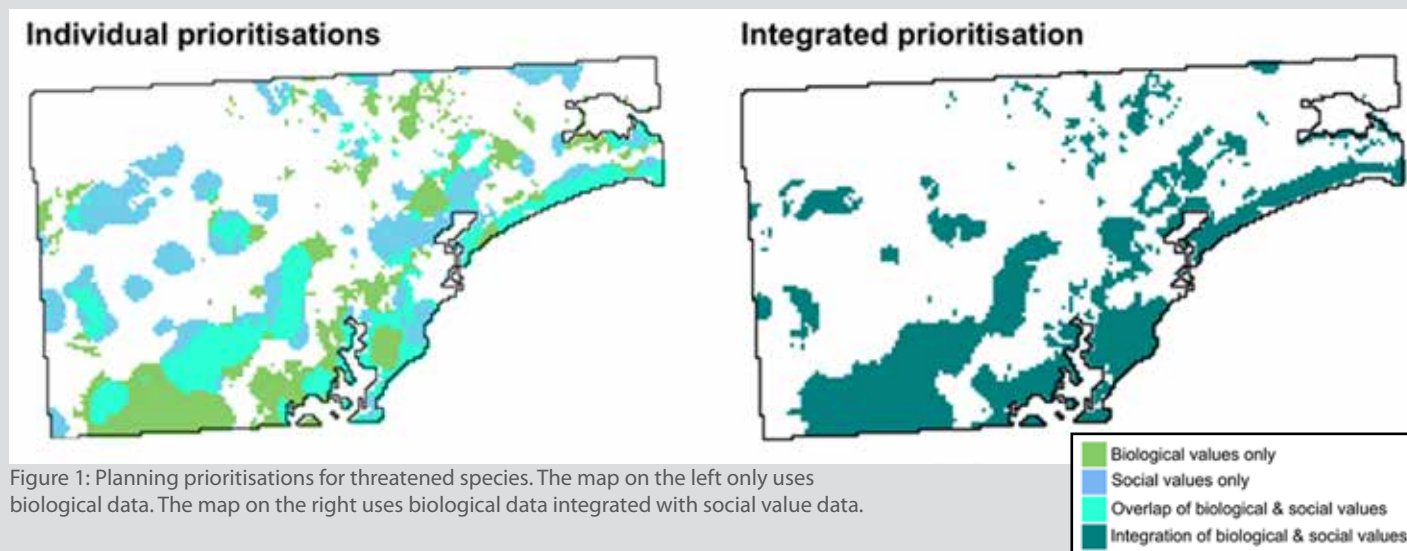
Integrating social values into spatial conservation planning

Understanding how society perceives and values different areas of the landscape is important for effective land-use planning.

they associated with each of the social values for biodiversity and development. The responses were then digitised and used to create density maps for each social value in the region.

In addition to this data on social values, the researchers used biological data to represent areas important for conservation. Seven fauna species considered to be vulnerable to land clearance were selected to represent the biological values in the Lower Hunter Valley. These species were mapped onto the landscape using species distribution models, derived from occurrence data.

Not surprisingly, the best option for conserving the seven threatened species included in the model was the distributions obtained when the prioritisation only considered biological data (Figure 1). Interestingly, however, a similar proportion of each species' current distribution was captured when biological and social values for biodiversity were integrated and prioritised together, although the spatial location of some conservation areas changed. Even when the areas perceived to be most important for development were forced out and the remaining sites were prioritised for both biological and social values for biodiversity, Zonation (the planning software used) still managed to find a solution that gave reasonable protection to the seven fauna species. This is good news for planners as it demonstrates spatial flexibility in the way conservation targets may be met in the Lower Hunter Valley.



Indeed, making use of social values is arguably one of the most important challenges in modern conservation planning, yet their potential remains poorly exploited.

Previous research suggests the inclusion of social values can reduce conflicts between stakeholders and enable a more efficient implementation of conservation actions. However, the potential trade-offs related to incorporating social values into spatial conservation planning are not well understood. These gaps in knowledge led Amy Whitehead and colleagues to investigate methods for integrating social values into spatial conservation planning.

Incorporating human and social dimensions into policy evaluation

Incorporating human and social dimensions into the evaluation of environmental policy options can improve the robustness of policy decisions.

Decision science provides the necessary tools for rational decision making under uncertainty and complexity (Raiffa, 1968). The discipline has evolved rapidly in recent years with Australian scientists leading theoretical developments and applications in environmental management and biodiversity

conservation. This is reflected by an increasingly significant role of decision science in conservation decisions in Australia. Sophisticated ecological and economic models are now available to help inform environmental decision making in the key areas of climate change adaptation (eg, Wintle et al, 2011), efficient monitoring strategies (eg, Wintle et al, 2010) and managing threats to species persistence (Nicholson et al, 2006).

While the sophistication of environmental decision models has improved, incorporating social science is still in its infancy (Milner-Gulland, 2012). This is an important limitation as understanding human demography, behaviour and socioeconomics is critical to understanding and managing ecological patterns and processes (Liu 2001).

More specifically, improving the effectiveness of complex policy instruments such as biodiversity markets requires a better understanding of human responses to incentives and disincentives because they determine the success or failure of such policies (Bekessy & Cooke, 2011). By failing to consider social science in environmental decision making we are at risk of choosing strategies that are unlikely to work.

Biodiversity offsets are a point in case. These are an increasingly popular policy mechanism whereby developers 'offset' a development by enhancing natural values elsewhere. But,

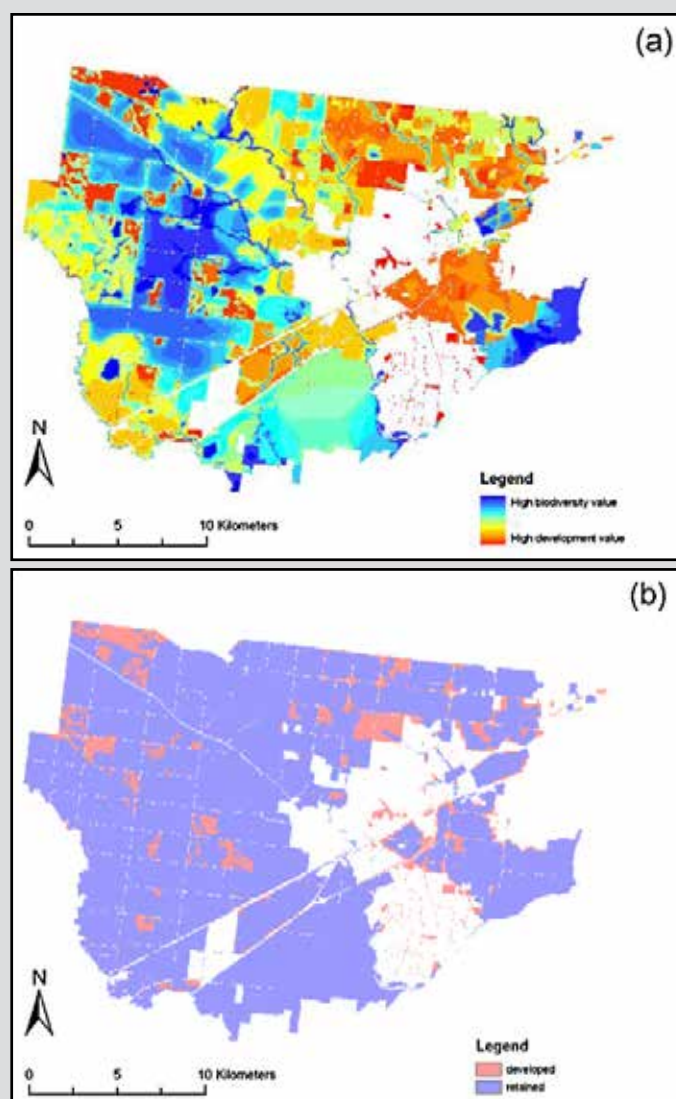
Planning for biodiversity and development in the urban fringe

Biodiversity near and within urban areas brings many benefits but its maintenance involves complex trade-offs between competing land uses. Sarah Bekessy and colleagues demonstrated how these trade-offs can be better described to facilitate more transparent, efficient and democratically derived urban planning (Bekessy et al, 2012).

They used reserve design tools in a novel way to identify priority development sites. The approach is based on a synthesis of ecological, social and economic data. Then trade-offs between biodiversity conservation and other key development objectives were quantified (eg, see fig 2). Other key development objectives included transport planning, flood risk and food production. And they demonstrated how all this can be done using a case study of changes in land use across the city of Wyndham, a local government west of Melbourne.

While this tool provides a transparent mechanism for articulating trade-offs in urban planning, it does not indicate whether decisions are ultimately 'acceptable'. The decision to clear habitat to meet competing objectives is a social one, but should be made acknowledging the risks to environmental and other concerns. A decision theory framework that articulates costs, benefits and risks could be useful in this context.

Figure 2. A map of the study area showing its biodiversity value (a). (b) shows areas representing the lowest ranked 10% of the landscape in terms of biodiversity value. If biodiversity was your only consideration when it came to development then the areas in pink are places you would develop. Of course, in the urban zone there are many other considerations such as proximity to transport and flooding risk. The researchers explored multiple scenarios where these other considerations are also factored in to demonstrate it is possible to optimise urban planning taking into account multiple considerations.



Offsets crowd out conservation volunteerism

Lots of us spend our spare time volunteering with various organisations to plant trees, control weeds, and generally doing good environmental work. What would you think if you found out that the work you were doing was actually part of a legally required offset for a large corporation? Or worse still – what if revegetation work you contributed to over many years was retrospectively exchanged for an equivalent amount of environmental damage by a developer? You'd probably be pretty annoyed – and maybe less likely to volunteer for environmental work in the future.

Sound far-fetched? Think again. In one recent infamous example, Canberra residents learned that their volunteer work to revegetate a local park was subsequently used to 'offset' the loss of nearby woodland for urban development. And, increasingly, funding is being made available to community groups to do restoration work for threatened species, like Carnaby's black-cockatoo, but the fine print reveals that the funding is part of an offset for habitat loss.

So, if people react to discovering they're doing work that will be negated by biodiversity impacts elsewhere to generate profit for someone else, they might just withdraw their volunteer labour, causing declines in environmental volunteerism and the benefits that it generates. This is referred to as 'crowding out'.

See [Decision Point #91](#) for a longer discussion on offset policies and perverse incentives.



Many people gladly give their time to community plantings. It's an opportunity to 'give back' to the environment. But what if the revegetation work was retrospectively exchanged for an equivalent amount of environmental damage by a developer? Would people still be happy about donating their time?

as Ascelin Gordon and colleagues discovered, a failure to consider social values in the policy formulation makes perverse outcomes much more likely (Gordon et al, 2015) (see the box on Offsets crowd out conservation volunteerism).

Psychological insights and social values

Engagement with psychologists and social scientists can improve our ability to anticipate responses to policies, including positive or negative feedback loops and perverse outcomes. Consider these examples.

Strategic framing of biodiversity in neoliberal terms may have unintended impacts on how people perceive and engage in biodiversity conservation. Consider the increasing use of utilitarian values to define nature. For example, there has been a dramatic increase in the use of the term 'ecosystem services' which is framed as the value of the benefits provided by nature, something that can have monetary values attached to it (see the box on a critique of ecosystem services as a communication strategy).

Kusmanoff et al (2017) analysed the language being used by environmental agencies and found that there has been a decrease in the use of the term 'biodiversity' and an increase in the use of economic language, including regular use of 'ecosystem services' concepts. In contrast, over the same time period, 'biodiversity' has increased in use within scientific literature. What does this mean for biodiversity conservation?

While this may reflect a strategic response by these agencies to better engage with both the general public and decision makers within what is an increasingly dominant neoliberal

paradigm, we argue it may also have unintended (possibly adverse) impacts on how people think about and engage with biodiversity conservation. There is concern that consistent framing of biodiversity in economic terms (such as ecosystem services) will promote the value of biodiversity as a resource over its intrinsic value.

Social values and psychological insights can also help in designing more effective environmental policy such as efforts to get more landowners to participate in biodiverse carbon plantings. Torabi et al (2016) sought to determine what drives landholders' participation in biodiverse carbon plantings? They found that the rate of landholder participation depends on many social and environmental drivers. The amount of management required and the landholder's value of biodiversity was in many cases more influential than financial incentives.

Values other than dollars was also important in understanding what make for a successful stewardship program over longer time frames (Selinske et al, 2016) and in maximising engagement with landholders engaged in private land conservation (Kusmanoff et al, 2016).

The way ahead for socio-ecological thinking

In terms of socio-ecological thinking, we're only just getting started. Over the seven years of CEED, the interdisciplinarity, sophistication and robustness of our approaches to embedding social complexity into decision science has grown substantially. We've moved from simple economic models of human responses to environmental interventions to much more nuanced understanding of how to represent values, predict

Strategic framing for landholder engagement

A study of how private-land conservation organisations frame the benefits of participation has found a bias for emphasising the environmental benefits, while under-emphasising the benefits to landholders and the wider social benefits (Kusmanoff et al, 2016). The researchers found that the success of these private-land conservation efforts is tied to the engagement of landholders, however only a small proportion of landholders participate in conservation.

The study analysed the websites of 20 of Australia's leading private land conservation schemes and categorised how the benefits of participation were framed; whether framed as benefits to landholders, to society or to the environment.

They found a predominance of environmentally-framed benefits. The lack of emphasis on social benefits in particular is a missed opportunity to engage community-minded landholders who don't necessarily identify with the conservation movement. By appealing to those people who are already receptive to conservation messages, we are only ever going to recruit from the same potential pool of landholders. But we can increase that pool beyond the conservation-minded, by emphasising both the social and personal benefits of conservation.

The researchers concluded that we must think strategically about who we are trying to reach, what motivates them, and how to frame our messages to better engage them.

how people may respond individually or collaboratively, and understand how to design messages and programs that are more likely to achieve desired behaviour changes.

Yet, we are really only getting started! This research space presents many exciting challenges, both theoretical and practical, and the demand from policy makers for socio-ecological thinking is only growing. A key theoretical challenge is how do we move from the rational optimizer to behavioural economics and beyond? How much social complexity versus ecological complexity should we include in our models?

We have identified judicious opportunities to advance research in socio-ecological analysis and modelling. This has been largely due to recent innovations in social research, especially in the area of behaviour change. For example, the field of behavioural economics (a partnership between economics and psychology) has significantly influenced the way that governments attempt to influence behaviour in many sectors, including health and taxation. Its application to the environment, however, has so far been minimal.

We can also draw on the innovative idea of 'experimentalism' from sociology, which adds value to the established approach of adaptive management, and the research field of 'political communication' from political science, which is concerned with production, dissemination, procession and effects of information within a political context.

These research fields have significant untapped potential to contribute to improved thinking and decision making about environmental management and policy. Importantly the new insights generated will allow for the design of environmental policies to achieve desired behavioural changes more efficiently

and reliably, and in ways that do not undermine existing pro-environmental behaviours and motivations.

Environmental decision science has come a long way in recent years in helping decision makers navigate an increasingly complex world. And yet the social and psychological dimensions of this complexity are only now being understood and acknowledged. The big challenge is to effectively incorporate these new learnings into the way we make decisions.

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Reference

- Bekessy SA & B Cooke (2011). Social and cultural drivers behind the success of Payment for Ecosystem Services (PES), In: *Payment for Ecosystem Services and Food Security*. Ottaviani, D & N El-Hage Scialabba (eds). Food and Agriculture Organization, Rome. pp. 141-155.
<http://www.fao.org/rio20/special-features/payments-for-environmentalservices/en/>
- Bekessy SA, M White, A Gordon, A Moilanen, MA McCarthy & BA Wintle (2012). Transparent planning for biodiversity and development in the urban fringe. *Landscape and Urban Planning* 108: 140-149.
For a discussion on this paper see [Decision Point #68](#)
- Bekessy SA, MC Runge, AM Kusmanoff, DA Keith & BA Wintle (2018). Ask not what nature can do for you: A critique of ecosystem services as a communication strategy. *Biological Conservation* 224: 71-74.
- Gordon A, WT Langford, L Bastin, AM Lachner & SA Bekessy (2013). Simulating the value of collaboration in multi-actor conservation planning. *Ecological Modelling* 249: 19-25.
For a discussion on this paper see [Decision Point #73](#)
- Gordon A, JW Bull, C Wilcox & M Maron (2015). Perverse incentives risk undermining biodiversity offset policies. *Journal of Applied Ecology* 52: 532-537 doi: 10.1111/1365-2664.12398
For a discussion on this paper see [Decision Point #91](#)
- Kusmanoff AM, MJ Hardy, F Fidler, G Maffey, C Raymond, MS Reed, JA Fitzsimons & SA Bekessy (2016). Framing the private land conservation conversation: Strategic framing of the benefits of conservation participation could increase landholder engagement. *Environmental Science & Policy* 61: 124-128.

A critique of ecosystem services as a communication strategy

Ecosystem services were devised in the 1970s to generate interest in biodiversity conservation. Since its inception as a communications tool in the 1970s, the concept of ecosystem services has become pervasive in biodiversity policy. While the goal of securing ecosystem services is absolutely legitimate, Bekessy et al (2018) argue that it has had limited success as a vehicle for securing public interest and support for nature, which is crucial to securing long-term social mandates for protection.

Emerging evidence suggests that focusing on ecosystem services rather than the intrinsic value of nature is unlikely to be effective in bolstering public support for nature conservation. Bekessy and colleagues found that positive messages of nature's aesthetic, cultural and spiritual aspects may be more beneficial. Communicators should think carefully about their audience when framing messages about nature

The importance of non-financial incentives for long-term stewardship

Changing human behaviour is fundamental to the success of conservation programs. Fostering an ethic of 'stewardship' on private land is one form of behaviour change increasingly being sought to protect key biodiversity areas. When planning private land conservation initiatives, multiple incentives are employed to attract landowners into short-term and long-term conservation contracts.

Given the long-term horizons for biodiversity conservation, Matthew Selinske and colleagues were curious to discover which incentives contribute to long-term stewardship (Selinske et al, 2016). Long-term outcomes are important given uncertain political support for conservation initiatives across many local- and national-level governments.

The researchers interviewed 113 landowners across three different program types and contexts: the Biodiversity Stewardship Program (a conservation contract program in South Africa); the Greenfleet biodiverse carbon-offset program in Victoria, Australia; and EcoTender, a reverse auction and covenanting program run by the Victorian State Government.

They found that financial incentives may help in increasing participation by reducing barriers to uptake (and to pay for management projects). However, for participants across these three programs, financial incentives were not a main driver to



A Biodiversity Stewardship Program landowner looking out over recently burned fynbos near Elgin, South Africa. (Photo by Matthew Selinske)

participate. They found over 90% of respondents participate because they care about their land and desire to restore and protect it (often in perpetuity). The mechanism that protects the land (eg, via a covenant or easement) becomes a main incentive to join. As one EcoTender participant put it: "Because [the restoration is] something I would have done anyway but I think the real bait for me was the covenant. If I did all this [work] and after I've gone somebody buys the land and knocks it all over, what's the point?"

Kusmanoff, M Alexander, F Fidler, A Gordon & S Bekessy (2017). Decline of 'biodiversity' in conservation policy discourse in Australia. *Environmental Science & Policy* 77: 160-165.

Liu J (2001). Integrating ecology with human demography, behavior, and socioeconomics: Needs and approaches. *Ecological Modelling* 140: 1-8.

Milner-Gulland EJ (2012). Interactions between human behaviour and ecological systems. *Philosophical Transactions of the Royal Society B*, 367, doi: 10.1098/rstb.2011.0175.

Nicholson E, MI Westphal, K Frank, WA Rochester, RL Pressey, DB Lindenmayer & HP Possingham (2006). A new method for conservation planning for the persistence of multiple species. *Ecology Letters* 9: 1049-1060.

Raiffa H (1968). *Decision Analysis: Introductory Lectures on Choices Under Uncertainty*. Addison-Wesley, Reading, MA.

Selinske MJ, B Cooke, N Torabi, MJ Hardy, AT Knight & SA Bekessy (2016). Locating financial incentives among diverse motivations for long-term private land conservation. *Ecology and Society* 22:7
For a discussion on this paper see [Decision Point #102](#)

Torabi N, L Mata, A Gordon, G Garrard, W Wescott, P Dettmann & S Bekessy (2016). The money or the trees: What drives landholders' participation in biodiverse carbon plantings? *Global Ecology and Conservation* 7: 1-11.
For a discussion on this paper see [Decision Point #98](#)

Whitehead AL, H Kujala, C Ives, A Gordon, PE Lentini, BA Wintle, E Nicholson & CM Raymond (2014). Integrating biological and social values when prioritizing for biodiversity conservation. *Conservation Biology* doi: 10.1111/cobi.12257
For a discussion on this paper see [Decision Point #80](#)

Wintle BA, SA Bekessy, DA Keith, BW vanWilgen, M Cabeza, B Schröder, SB Carvalho, A Falcucci, L Maiorano, TJ Regan, C Rondinini, L Boitani & HP Possingham (2011). Ecological-economic optimization of biodiversity conservation under climate change. *Nature Climate Change* 1: 355-359.

Wintle BA, MC Runge & SA Bekessy (2010). Allocating monitoring effort in the face of unknown unknowns. *Ecology Letters* 13: 1325-1337.



How much does the community value nature? Is it enough to simply install cut outs of kangaroos in peri-urban developments to give people their daily nature fix?

A very short history of *Decision Point*

Decision Point was established in 2008 by the Applied Environmental Decision Analysis Hub (AEDA: 2007-2010, through funding from the Commonwealth Environment Research Facility), the decision science network that preceded CEED. The magazine was also co-funded by the National Environmental Research Program Environmental Decisions Hub (NERP ED: 2011-2015). Over its life-time, CEED has been the major funder of *Decision Point* (2011-18).

In its first year *Decision Point* had a circulation of a few hundred – mainly researchers, policy makers, resources managers and members of the general community. It found favour with its audience and its circulation has steadily grown to be around 7,000 subscribers.

Having chalked up over 100 issues, *Decision Point* has contributed to a cultural shift in policy formulation in Australian governments (at multiple levels). That shift has been away from ad hoc, opaque decision making (in regards to policy relating to nature, environment and threatened species management) towards more transparent, accountable, systematic and adaptive decision making.

Over the course of *Decision Point*'s life, the field of environmental decision science has grown from a little utilised academic pursuit to become an important element of environmental policy. *Decision Point* has played a role in this transformation.

Back issues of *Decision Point* will still be available on an archived website (<http://decision-point.com.au/>) but no new issues are currently being planned beyond 2018.

A few quotes from readers

Here's a bit of feedback from readers. We've selected quotes reflecting the breadth of our readership, from 'coal-face' conservation managers to the head of the Australian Department of the Environment.

"Decision Point was one of the best things to come out of the NERP Program. Among other things, it provides a direct and timely link from the work of researchers to policy advisers and policy makers."
Paul Grimes, Former Secretary of the Australian Government Department of Sustainability, Environment, Water, Population and Communities.

"Just wanted to commend you guys on an excellent mag – we use it to inform a lot of our thinking."
Todd Maher, Natural Resources Commission, NSW Government

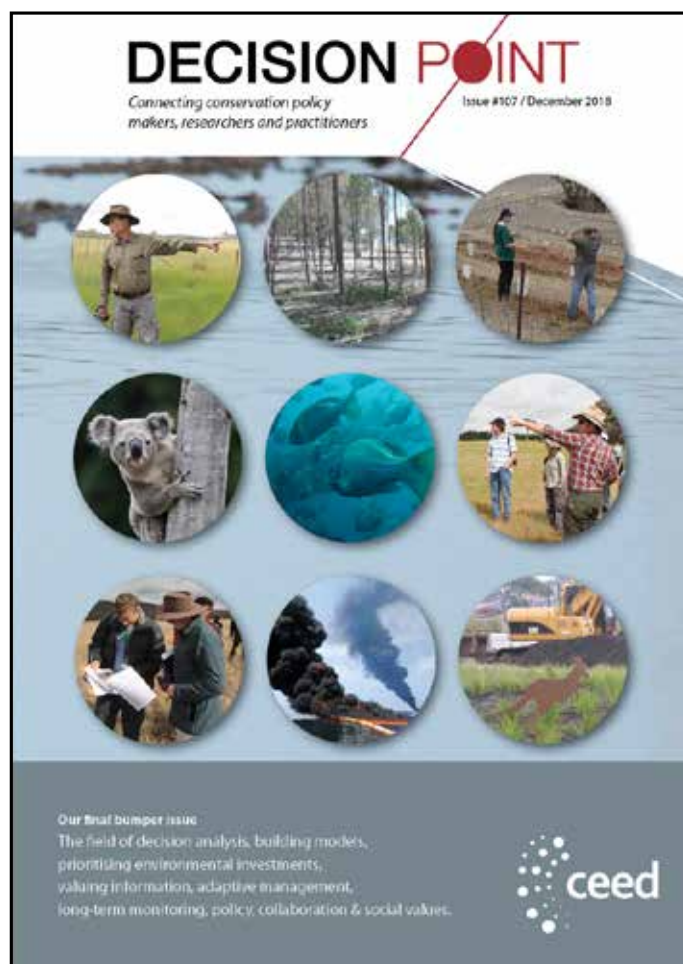
"It's of great benefit being able to tap into the current best practice approaches and ideas emerging from academia."
Julian Seddon, Dept. Environment and Sustainable Development, ACT Government

"Decision Point would have to be the most relevant and insightful publication I receive. The coverage of issues is excellent, the content innovative and the research findings are incredibly useful in my everyday work."
Kirsti Sampson, Southern Rivers CMA, NSW

"Would like to do something similar to Decision Point for ecosystems and biodiversity in South Africa - this is a lovely example - well done."
Wiida Basson, Senior Communicator, Council for Scientific and Industrial Research, South Africa

"Good information, relevant pitch, stimulating debate and provocative essays (should be more of it)."
Peter Copley, Senior Ecologist, Threatened Species & Ecological Communities Unit, South Australian Government

"Frankly, Decision Point is one of the only newsletters I receive that I prioritise reading, and I do this because the content is captivating, and the format makes it easy to understand and digest the huge amount of information."
Andrew Chin, Great Barrier Reef Marine Park Authority



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