Marine Biodiversity Hub

National Environmental Science Programme

Review of decision support tools and their potential application in the management of Australian Marine Parks

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Project B1 Road testing decision support tools via case study applications

December 2019

Milestone 1, RPv1 (2015)

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Preferred Citation

Walshe, T., Dempster, F., Pascoe, S. and Jennings. S. (2019). Review of decision support tools and their potential application in the management of Australian Marine Parks. Report to the National Environmental Science Program, Marine Biodiversity Hub. *Australian Institute of Marine Science.*

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Acknowledgement

This work was undertaken for the Marine Biodiversity Hub, a collaborative partnership supported through funding from the Australian Government's National Environmental Science Program (NESP). NESP Marine Biodiversity Hub partners include the University of Tasmania; CSIRO, Geoscience Australia, Australian Institute of Marine Science, Museums Victoria, Charles Darwin University, the University of Western Australia, Integrated Marine Observing System, NSW Office of Environment and Heritage, NSW Department of Primary Industries.

Important Disclaimer

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EXECUTIVE SUMMARY

The purpose of this report is to provide an appraisal of the strengths and weaknesses of a suite of decision-support tools for the kinds of problems encountered by marine park policy-makers and managers.

In organisations around the world, the process by which many decisions are made is unstructured. The most common method of organisational decision-making is through open dialogue in a committee setting. This may be entirely adequate for the many problems that involve small consequences, but it is unlikely to be appropriate where the stakes are high. Even where detailed information and analyses are marshalled to support the committee meeting process, unstructured conversation is prey to the frailties of groupthink, deference to authority, and a bias towards retaining the status quo. Meetings typically exceed the cognitive limits of the human brain. Psychologists have clearly demonstrated that our minds are incapable of processing more than about seven things at any one time. A committee discussion typically involves dozens of things, including issues, alternatives, pros, cons, objectives and criteria.

To the extent that they capture sound logic, formal decision support tools have advantages over unaided decision-making. Apart from buffering against cognitive limitations and negative group dynamics, a documented and traceable protocol will encourage decision-makers to be clear about judgments and assumptions.

Many predictive tools and models, often based on empirical observation, provide partial decision support. They account for the consequences of a set of alternative policies or actions. Formal decision support tools go beyond the empirical estimation of consequences to address the development of creative alternatives for dealing with a problem, wrestling with trade-offs, coping with uncertainty, and identifying optimal solutions for an individual decision-maker or acceptable solutions for multiple co-managers or stakeholders. Specific tools vary in their emphases on these and other elements. This report describes a suite of tools and techniques that structure decision elements in ways that promote improved outcomes consistent with organisational objectives, focusing on management needs for Australian Marine Parks (AMPs).

Through recently released management plans for AMPs, Parks Australia have articulated core objectives and listed key values, pressures and potential actions. Although there is a dearth of information describing many values and pressures in any detail, and the effectiveness of candidate actions in some instances requires further evaluation, Parks Australia is now in a position to begin to utilise the tools and techniques of structured decision-making.

Section 1 of this report is targeted at AMP managers. It is a primer on decision-making in marine parks, highlighting the imperative for decision-making under uncertainty and how to go about it.

Section 2 makes up the bulk of this report. It is a compendium of tools and techniques that can be deployed in two broad classes of decisions:

Acceptable risk decisions – for example, routine tactical decisions faced by *Assessments and Authorisations* to allow or not allow an activity. Decision support tools for this class of problem include:

- Qualitative risk assessment
- Quantitative risk assessment
    - Logic trees
    - Bayesian Belief Networks
    - Monte Carlo simulation

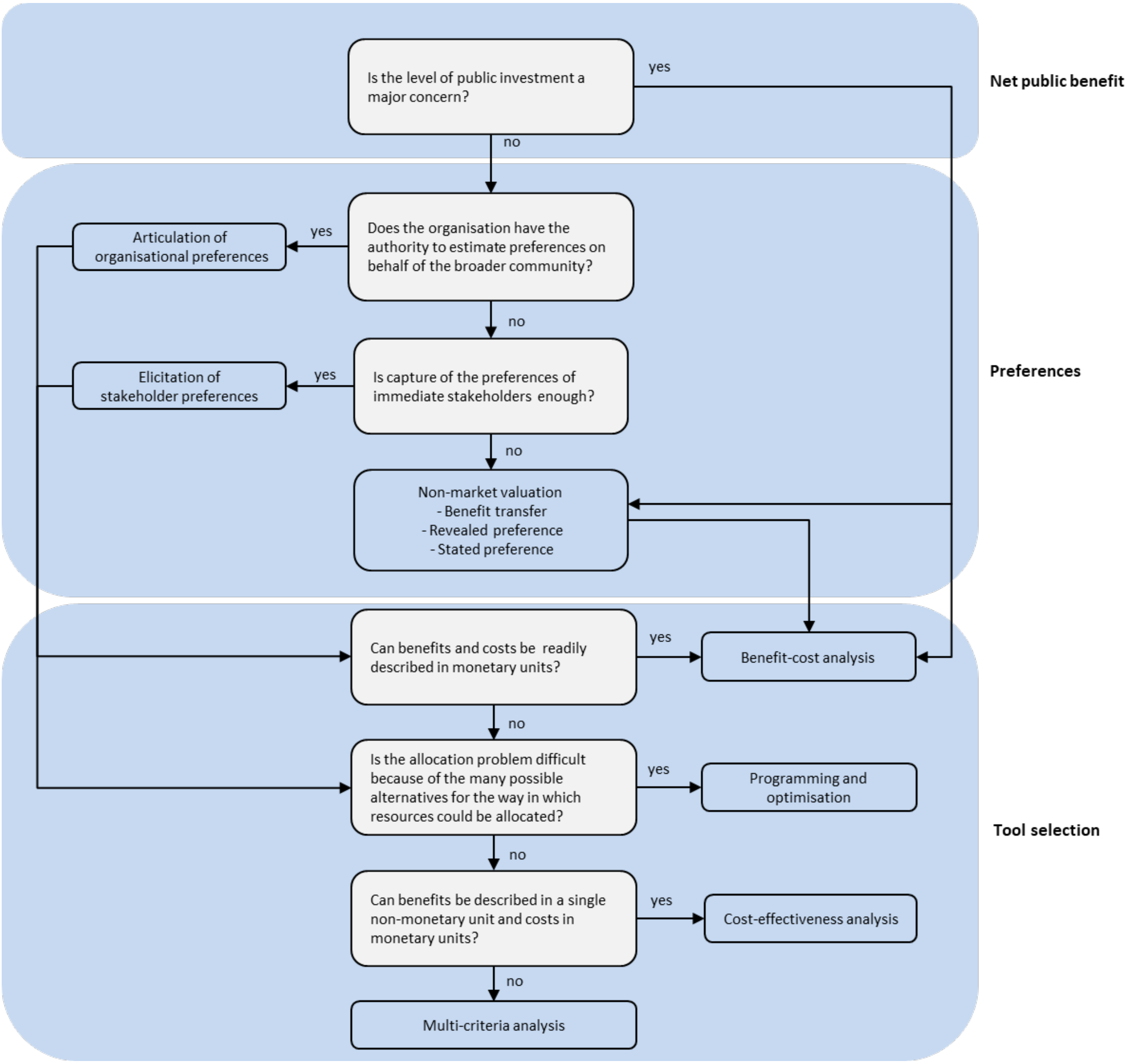
Essentially, this list represents a continuum from a simpler approach (qualitative risk assessment), to intermediate tools requiring some understanding of probabilistic reasoning (logic trees), to advanced and rigorous methods (Bayesian Belief Networks and Monte Carlo simulation).

Resource allocation across various candidate management actions, for example, the protection of many conservation values involves management of biophysical processes and human behaviour where allocation of effort across multiple program areas may be required. Allocation of finite resources in a way that maximises desired management outcomes, including non-monetary (non-market) values such as biodiversity or social equity, is a common type of decision conundrum. The decision tree below provides coarse guidance on circumstances in which various tools may be more applicable.

Some resource allocation decisions may pertain only to a single objective (e.g. prioritising investments in biodiversity conservation), while others require consideration of trade-offs across multiple objectives (e.g. conservation, cultural values, enjoyment and use). At broader scales, periodic review of management arrangements, scheduled to occur at years 9 and 10 in the life of the current management plans, could include a major multi-objective spatial planning exercise that is essentially a resource allocation problem. Decision support tools for this class of problem include:

- Cost-effectiveness analysis
- Benefit-cost analysis
- Multi-criteria analysis
- Qualitative management evaluation
- Programming and optimisation
- Viability analysis

We emphasise that the tool used in any specific context should be guided by the anticipated obstacles to an effective and enduring outcome. Sometimes those obstacles pertain to insufficient scientific understanding, or a reluctance to entertain creative alternative solutions, or difficulty in articulating and capturing stakeholder concerns. It may also be related to limited resources, time or capacity in which to engage with tools and / or make decisions. Often, the difficulty may centre around unpalatable trade-offs. Section 2 includes a substantial exploration of the role of preferences in dealing with trade-offs, and a set of approaches to dealing with trade-offs that vary in the extent to which they privilege the perspectives of an individual decision-maker, key stakeholders or the broader perspectives of society.



Logic tree to assist in the selection of an approach to resource allocation problems involving non-market values. The chosen path will vary from one decision context to another.

Much of the content of Section 2 is technical. It provides a resource for AMP managers and decision support providers to use in navigating difficult decisions. Some of the simpler tools will be accessible to managers today. For example, qualitative risk assessments with tailored context-specific consequence tables are used routinely in many organisations for acceptable risk decisions. The combined use of logic trees and cost-effectiveness analysis for resource allocation problems (see the illustrative example in section 1.3) requires only a basic understanding of probability theory. Others may be used to inform approaches that could be used by consultants or researchers engaged to assist with more demanding problems. A brief summary of each tool is tabulated below.

Summary of common tools and applications for marine park managers dealing with acceptable risk problems.

Tool	Main strength(s)	Main weakness(es)	Example application
Qualitative risk assessment	Ease of use.	Language based ambiguities that invite arbitrary error in assessments.	Management priorities for marine parks (Carey et al. 2007).
Quantitative risk assessment - Logic trees	Simple, visually accessible models.	Can become messy when used for complex problems.	Conservation status of ecosystems (Keith et al. 2013).
Quantitative risk assessment - Bayesian Belief Networks	Accounting for uncertainty and conditional relationships between system variables.	Large requirements for data and/or expert judgment.	Assessment of marine offsets (Jennings et al. 2015).
Quantitative risk assessment – Monte Carlo simulation	Accounting for uncertainty and change over time.	Large requirements for data and/or expert judgment.	Ecosystem modelling (Fulton et al. 2014).

Summary of common tools and applications for marine park managers dealing with the capture of preferences.

Tool	Main strength(s)	Main weakness(es)	Example application
Market values approaches	Best approach when markets are open.	Market distortions can lead to bias.	Habitat productivity (McArthur and Borland 2006).
Revealed preference approaches	Able to use prices to estimate some non-market values.	Requires strong analytical skills.	Valuation of dive-based tourism (Pascoe et al. 2014b).
Stated preference approaches	Best approach for non-market values for which revealed preference techniques are unavailable.	Requires sound survey design and strong analytical skills. Cost of administrating survey.	Community valuation of conservation assets of Ningaloo (Rogers 2013).
Benefit transfer	Cheap.	Poor translation from previous studies to current context.	Value of beaches to tourism (Raybould and Lazarow 2009)

Tool	Main strength(s)	Main weakness(es)	Example application
Deliberative and other non-monetary valuation	Stakeholder engagement.	Stakeholder preferences may not reflect broader societal preferences.	Prioritisation of marine values (Ogier and McLeod 2013)

Summary of common tools and applications for marine park managers dealing with resource allocation problems.

Tool	Main strength(s)	Main weakness(es)	Example application
Cost-effectiveness analysis	Simple to use.	Cannot directly inform circumstances where *status quo* or do-nothing arrangements are best.	Evaluation of alternative control measures for oil spills (Vanem et al. 2008).
Benefit-cost analysis	Most rigorous approach.	Typically requires strong analytical skills and considerable time and resources.	Evaluation of marine protected areas (Rees et al. 2013).
Multi-criteria analysis	Stakeholder engagement	Stakeholder preferences may not reflect broader societal preferences.	Fisheries management (Pascoe et al. 2013).
Programming and optimisation	Able to explore vast numbers of alternatives	Constraints can make trade-offs difficult or opaque.	Fisheries management (Dowling et al. 2011).

In Sections 3 and 4 of the report we explore impediments to the adoption of decision-support tools and provide tentative guidance on the circumstances in which one technique or approach may be more appropriate than others (see summary table and figure below). We note that tool selection is in part shaped by the skills of available personnel and by political considerations, such that processes or techniques that engender trust may be preferred over those that emphasise technical rigour.

In the immediate future, we suggest Parks Australia might usefully concentrate the development of in-house competencies in (a) analyses underpinning routine decisions for which the organisation has clear authority, and (b) accessing appropriate expertise for more challenging decisions, especially those that may compromise standing and trust because of the need to confront difficult trade-offs.

1. A PRIMER ON DECISION-MAKING FOR MARINE PARKS

Parks Australia's management plans for its marine estate (Director of National Parks 2018a,b,c,d,e) give effect to its organisational priorities via articulation of objectives. The objectives of the plans are to provide for:

a. the protection and conservation of biodiversity and other natural, cultural and heritage values of marine parks; and
b. ecologically sustainable use and enjoyment of the natural resources within marine parks, where this is consistent with objective (a).

Management programs and actions aligned with these objectives are administered by seven program areas within Australian Marine Parks (AMP). Specifically:

- Communication, education and awareness
- Tourism and visitor experience
- Indigenous engagement
- Marine science
- Assessments and authorisations
- Park protection and management
- Compliance

The primary mechanism by which Parks Australia delivers its dual objectives for AMPs of (a) protection and conservation, and (b) use and enjoyment, is through a zoning system and accompanying rules, delineated and applied throughout marine networks and parks. After protracted debate, the current zoning configuration was resolved via an extensive consultation process (Buxton and Cochrane 2015, Beeton et al. 2015, Director of National Parks 2017a,b,c,d,e). The process made use of scientific knowledge wherever it was available, but it is broadly recognized that, relative to terrestrial reserves, the understanding of values, pressures and the effectiveness of many management actions in AMPs is poor. Decision-making in these circumstances is plainly difficult. Parks Australia is committed to incrementally improving its knowledge base and decision-making over time (Director of National Parks 2018f). The tools and techniques presented in this report include some that could be deployed today and others that require a greater knowledge base and greater levels of technical proficiency. The report is a compendium of approaches that can be considered as knowledge and skills progressively improve.

Two broad classes of decisions capture many of those encountered by program areas:

- **Acceptable risk decisions** – for example, routine tactical decisions faced by *Assessments and Authorisations* to allow or not allow an activity.
- **Resource allocation** across various candidate management actions – for example, the protection of many conservation values involves management of biophysical processes and human behaviour. Thoughtful allocation of effort across multiple program areas, including *Park protection and management, Communication, education and awareness, Assessments and authorisations* and *Compliance* may be required. Some resource allocation decisions may pertain only to a single objective (e.g. prioritising investments in biodiversity conservation), while others require consideration of trade-offs across multiple objectives (e.g. conservation, cultural values, enjoyment and use). At broader scales,

periodic review of zoning arrangements represents a major multi-objective spatial planning exercise that is essentially a resource allocation problem.

Section 2 of this report deals in considerable detail with these two classes of decision problems. It provides a resource for AMP managers to use in navigating difficult decisions. Some of the simpler tools will be accessible to managers today. Others may be used to inform approaches that could be used by consultants or researchers engaged to assist with more demanding problems.

1.1 A general framework for thinking about decisions

Many routine small-stakes decisions are made by individual officers or unstructured exploration of key decision elements in a committee setting. For decisions with more substantial consequences, structured approaches may be more appropriate. In this section we offer a broad overview and key concepts on how to formally approach decision-making using structured approaches.

Structured decision-making is designed to aid logical and transparent decision making. It describes both the process of deconstructing decisions into various common components, and the broad set of tools used (see Figure 1 below).

A generic framework for structured decision-making is PrOACT, an acronym describing key components:

- **Pr**oblem formulation
- **O**bjectives
- **A**lternatives
- **C**onsequences
- **T**rade-offs.

Almost all decisions and formal decision processes involve these elements but vary in their emphases and treatment.

After the need for a decision becomes apparent via some event or trigger, *problem formulation* is about setting the context, clarifying who is the decision-maker, the role of stakeholders, and delineating what's in and out of scope in the decision. It often involves consideration of hard constraints, typically defined by strict statutory requirements, and considerations shaped by policy directions.

All decisions have at least one *objective* and at least two *alternatives*. Among the most straightforward (although not necessarily easy) are routine tactical regulatory decisions that involve a well-defined threshold for acceptable risk on a clear organisational objective (e.g. minimum distances to be maintained by whale watching tourism operators) and two alternatives – approve or disapprove. Among the most complex are periodic strategic reviews of zoning in marine parks, where there are many objectives reflecting multiple conservation, cultural and use values and potentially thousands of candidate alternatives for the spatial configuration of zones.

Consequences are the predicted performance of each alternative against each objective. In higher stakes decisions, they may involve sophisticated biophysical models and their outputs. For example, the consequences of a change in blue zones and green zones need to be estimated against objectives describing conservation outcomes and fisheries outcomes. *Trade-offs* are plainly a part of complex multi-objective zoning decisions. The extent to which a gain (or loss) in conservation is compensated by a loss (or gain) for fisheries is a value judgment that will vary from stakeholder to stakeholder. Acceptable trade-offs are shaped by notions of acceptable risk and risk attitude. In marine park settings, the precautionary principle advocates risk aversion when conservation values are at stake. Sometimes decisions are linked. Decisions made to promote recreation may limit options for extractive use or conservation.

But not all problems require formal treatment of trade-offs. Single objective regulatory decisions are an example. Likewise, the allocation of resources under a single objective (e.g. minimise non-compliances) need not invoke trade-offs if the budget is fixed, that is where someone else has decided how much resourcing should be made available. Trade-offs may be limited to the articulation of organisational preferences, or may include capture of stakeholder and broader societal views (see section 2.2.1)

The extent to which the various components of the PrOACT framework are emphasized in a decision or a decision-support tool should be shaped by the nature and challenges of the decision problem. Sometimes the capture of stakeholder-relevant objectives might be the key obstacle to effective outcomes. In other settings, the need for creative alternatives, or a means of navigating an overwhelming number of potential alternatives may be the stumbling block. Science-based organizations can fall into a trap of over-emphasizing the technical characterisation of consequences for a small subset of decision elements, believing that good science or good models alone will bring clarity. Much has been written about interpreting data, building predictive models and eliciting expert judgment for the estimation of consequences. This report gives scant treatment to these topics. Instead it introduces a suite of tools and techniques that focus more on the treatment of objectives, alternatives, and trade-offs. It offers coarse guidance on tool and technique selection (see section 4.1), but ultimately these choices need to be made by organisations and their perceived needs and analytical capacities.

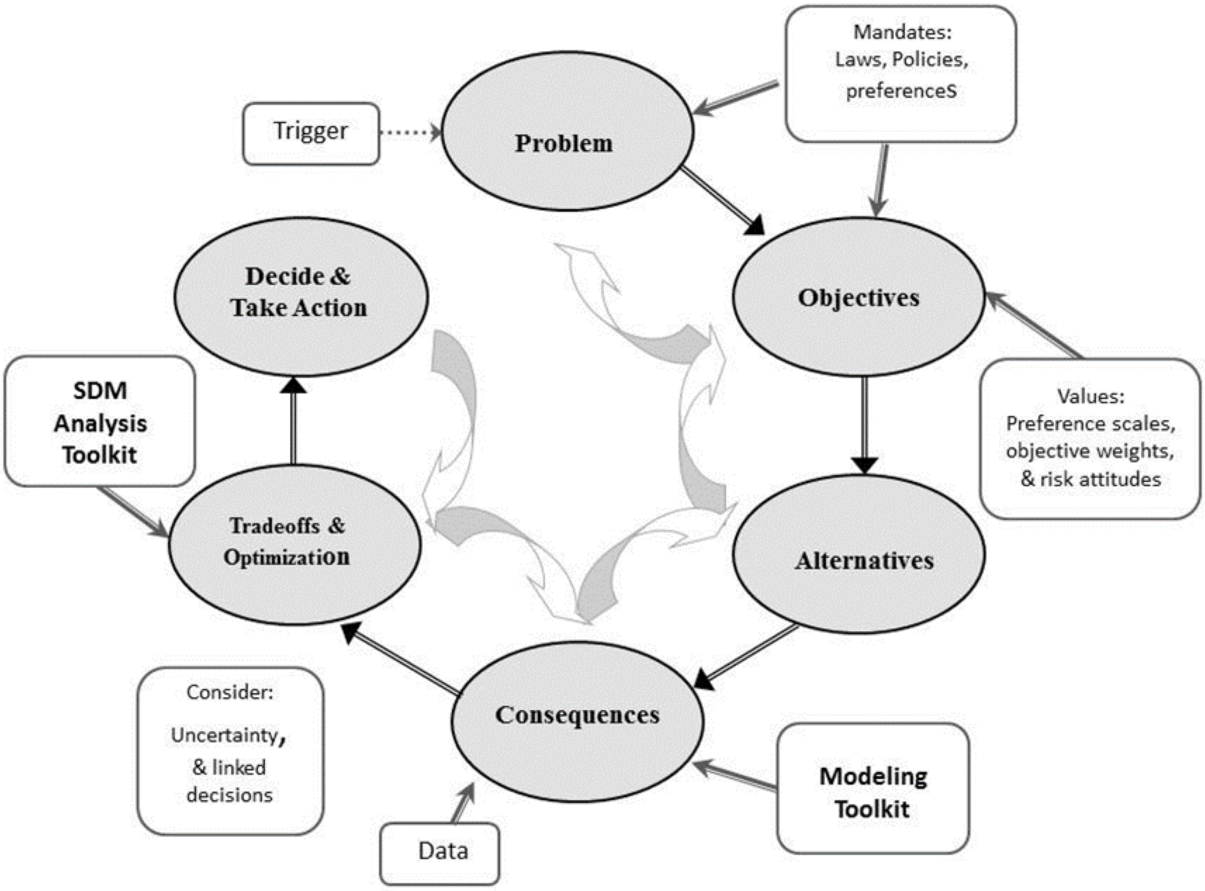


Figure 1. The structured decision-making framework. Source: Garrard et. al. (2017).

To introduce key concepts in structured decision-making and to highlight some of the content in the body of this report, we now explore two decision problems that may be encountered by Parks Australia in AMPs.

1.2 Decision-making under uncertainty

Parks Australia emphasises evidence-based decision-making in its Science Direction Statement and AMP management plans (Director of National Parks 2018a,b,c,d,e,f). While decision-making would be much easier if we had perfect or near-perfect knowledge of the consequences of candidate management actions, we're unlikely to be in such a position any time soon. Decisions need to be made while our understanding of values, pressures and the effectiveness of alternative actions matures.

A very common decision problem for park managers is whether or not to act (e.g. introduce stricter controls) on the basis of weak evidence suggesting decline in a species or ecosystem. Well intentioned conservation-minded stakeholders may overstate the evidence in vociferous appeals to act in the interests of the species or ecosystem. Against this background, managers need to make a decision. Their objectives include (a) protecting the species or ecosystem, and (b) avoiding unnecessary expense to the organisation (and taxpayers).

We can characterize the decision and its uncertainty in a logic tree (Figure 2; see also section 2.1.2). The square represents the decision point - act or don't act. Circles represent uncertainties. We don't *know* if the species is in decline or stable, we have only equivocal evidence there may be decline. Beyond status and trend of the species or ecosystem, the uncertainty if we do act may include speculation that the action itself may or may not be effective. The triangles at the ends of the branches represent consequences of each action under each uncertain state. The decision-maker wants to avoid the ecological costs of decline and the (possibly unnecessary) costs of implementing action. The decision-maker is in a pickle. She could delay until a better evidence base becomes available, but delay may worsen the (speculative) decline. And the acquisition of the evidence itself may be a non-trivial cost to the organisation.

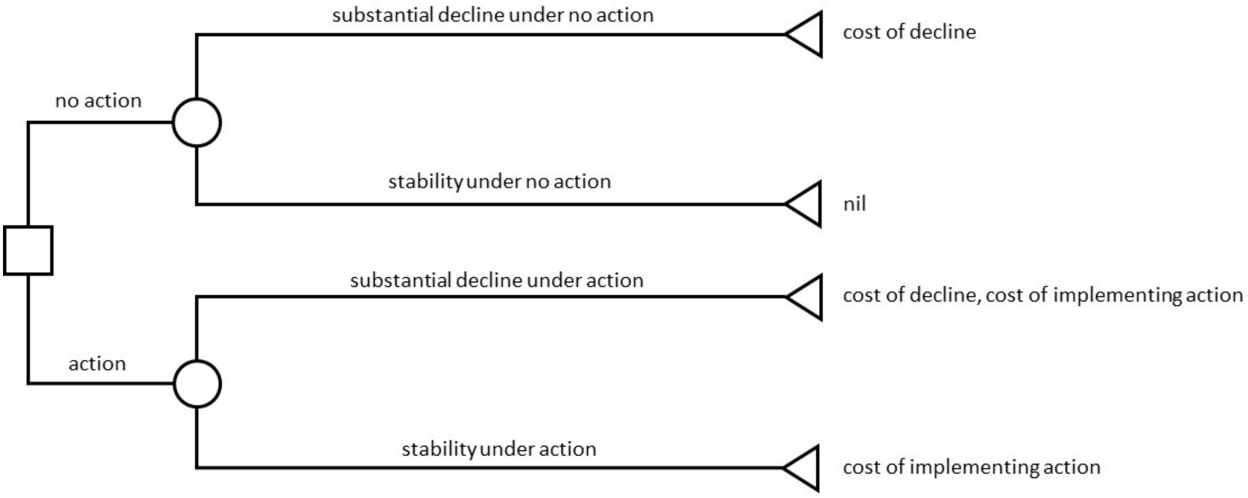


Figure 2. Logic tree for the decision on whether or not to act under speculative species or ecosystem decline.

For individual decisions, uncertainty inevitably implies we will sometimes make the wrong call. But if we can deal with uncertainty coherently, we can make rational decisions that best serve the organisation's objectives in the many decisions made over the long run. If we can assign probabilities to our uncertain nodes and quantify the consequences, we are on our way. Let's say our heroic decision-maker makes the following judgments:

- Under the *no action* alternative, the weak evidence base implies about a 0.70 probability of decline.
- Under the *action* alternative, there remains a 0.10 probability of decline because the effectiveness of the action is not entirely proven.
- The cost of implementing the action is $5 million.
- The ecological costs of decline (if true) are difficult to gauge, but let's say our decision-maker hazards a guess that the costs would be about 40 times greater than the cost of implementing the action, or in monetary terms, about $200 million.

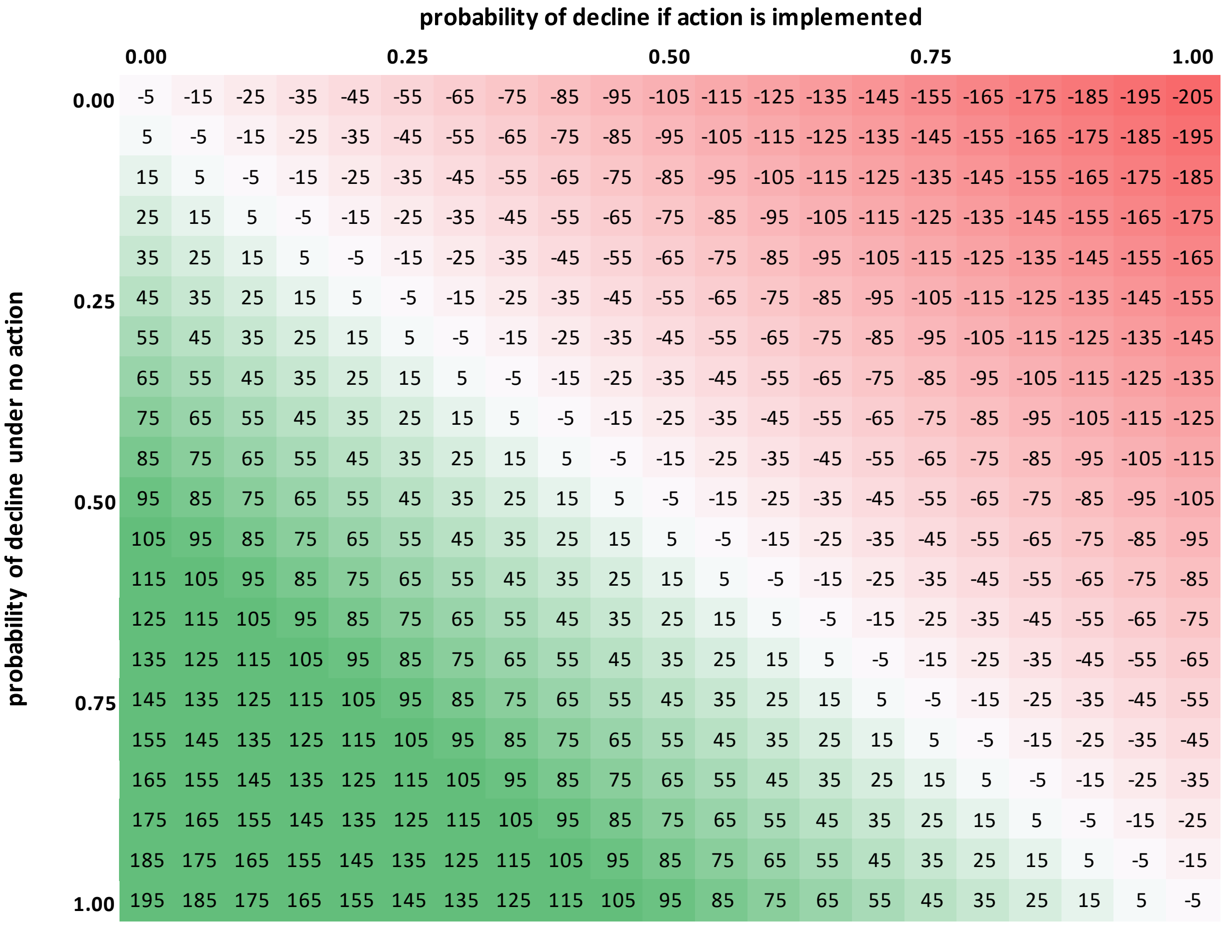
Now we can calculate the expected (i.e. probability weighted) consequences of the two alternatives:

No action = 0.70 × -$200M + 0.30 ×$0M = -$140M
Action = 0.10 × -$205M + 0.90 × -$5M = -$25M

The best action is the one with the highest expected consequence (i.e. least cost), which in this case is to implement the action at $25M.

Probabilistic judgments are tough to make. We rarely have a crisp sense of what probability to assign in any setting, even those where data are relatively plentiful. Figure 3 shows the outcomes of calculations for expected consequences under the full range of possible probabilities for our two uncertainties. The numbers in the body of the figure report the difference between the expected consequences of action and no action, at probability increments of 0.05. For our example above, where we used 0.70 for the probability of decline under no action and 0.10 under action, the value is the difference in expected consequences, -25 - -140 = 115. Where values are positive (shaded green) the *action* alternative is better than no action, and negative (red) when no action is the rational choice. The degree of shading indicates the conviction with which we would act or not act. Unsurprisingly, we have a clear argument for acting where the probability of decline under no action is high and we believe our action is highly effective such that it reduces the risk of decline to zero or near-zero (i.e. the bottom left hand corner of Figure 3).

Figure 3 can be used to make coarse decisions when we’re unsure what probabilities to assign. For instance, we may think decline under no action is plausible, but not likely, and could range anywhere from 0.25 to 0.50. We may be confident in the effectiveness of our action, but not entirely so, with probability of decline if action is implemented being anywhere from zero to 0.25. Within these two ranges almost all the cells in Figure 3 are positive, indicating *action* to be the better choice. Note that despite being less than 50% convinced of decline, the best choice can still be to proceed with action implementation. In our example, it works out this way because the ecological costs of decline ($200 million) are much greater than the costs of implementation ($5 million).



probability of decline if action is implemented

probability of decline under no action

	0.00					0.25					0.50					0.75					1.00
0.00	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155	-165	-175	-185	-195	-205
	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155	-165	-175	-185	-195
	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155	-165	-175	-185
	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155	-165	-175
	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155	-165
0.25	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145	-155
	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135	-145
	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125	-135
	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115	-125
	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105	-115
0.50	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95	-105
	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85	-95
	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75	-85
	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65	-75
	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55	-65
0.75	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45	-55
	155	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35	-45
	165	155	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25	-35
	175	165	155	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15	-25
	185	175	165	155	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5	-15
1.00	195	185	175	165	155	145	135	125	115	105	95	85	75	65	55	45	35	25	15	5	-5

Figure 3. Circumstances in which it is better to act (green) or not act (red) according to probabilistic judgments of ecological decline in the presence and absence of management action, and where the costs of implementing the action is 5 and the cost of decline is 200 (in arbitrary units).

Of course, we're also often uncertain about the quantities we should assign to costs (especially ecological costs) together with probabilities (see section 2.2.1 on valuation of non-market consequences). Figure 4 shows nine combinations of values for ecological costs and costs of action implementation. As the costs of implementation rise relative to the costs of ecological decline, the 'zone' in which action is rational becomes progressively restricted to circumstances where we are highly convinced of decline and of the effectiveness of our action.

To this point we've explored only two alternatives: act or don't act. Often under uncertainty, the natural impulse of the decision-maker is to delay until more evidence is accumulated and the right choice becomes apparent. If our objectives are to minimize the adverse consequences of ecological decline and costs to the organization, then only sometimes delay and learning through research, monitoring or adaptive management is rational. Often it is not (McDonald-Madden et al. 2010).

Figure 5 extends the logic tree in Figure 2 to include the 'delay and learn' alternative. The calculation for expected consequences (not shown) are a little more involved, but not difficult(see section 2.2.3). Outcomes for the scenario where costs of implementation are $5million and costs of ecological decline are an estimated $200 million are shown in Figure 6. The values in the body of Figure 6 report the probability weighted value of learning. They can be interpreted as upper bounds on the price a rational decision-maker should pay for perfect knowledge regarding the status of the species or ecosystem *and* the effectiveness of the candidate management action. Values shaded green represent circumstances where delay and learning are a good option. Yellow and orange cells indicate a lesser conviction in the merit of delay. And red cells show circumstances where generally the decision-maker is likely to be better off committing to a decision today, despite uncertainty (the exception is where information gathered through research or monitoring is inexpensive to acquire). Perhaps intuitively, the value of learning is greatest near the diagonal, representing probability combinations where the rational choice under uncertainty changes from act to don't act (see Figure 3).

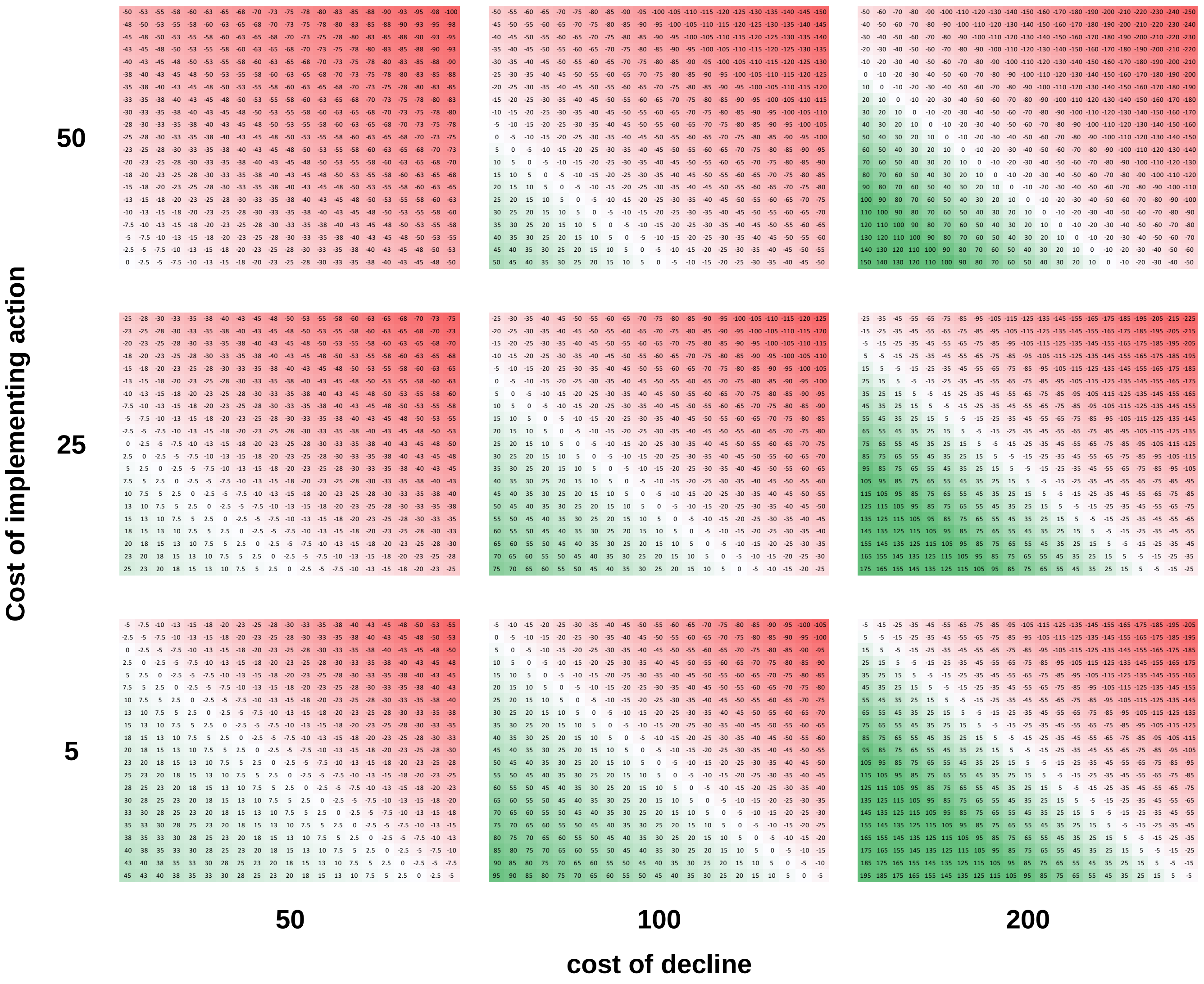


Figure 4. Circumstances in which it is better to act (green) or not act (red) according to probabilistic judgments of ecological decline in the presence and absence of management action, for nine combinations of implementation cost and ecological cost. For each combination, the probability of decline if action is implemented is the x-axis, and the probability of decline under no action is the y-axis, consistent with Figure 3.

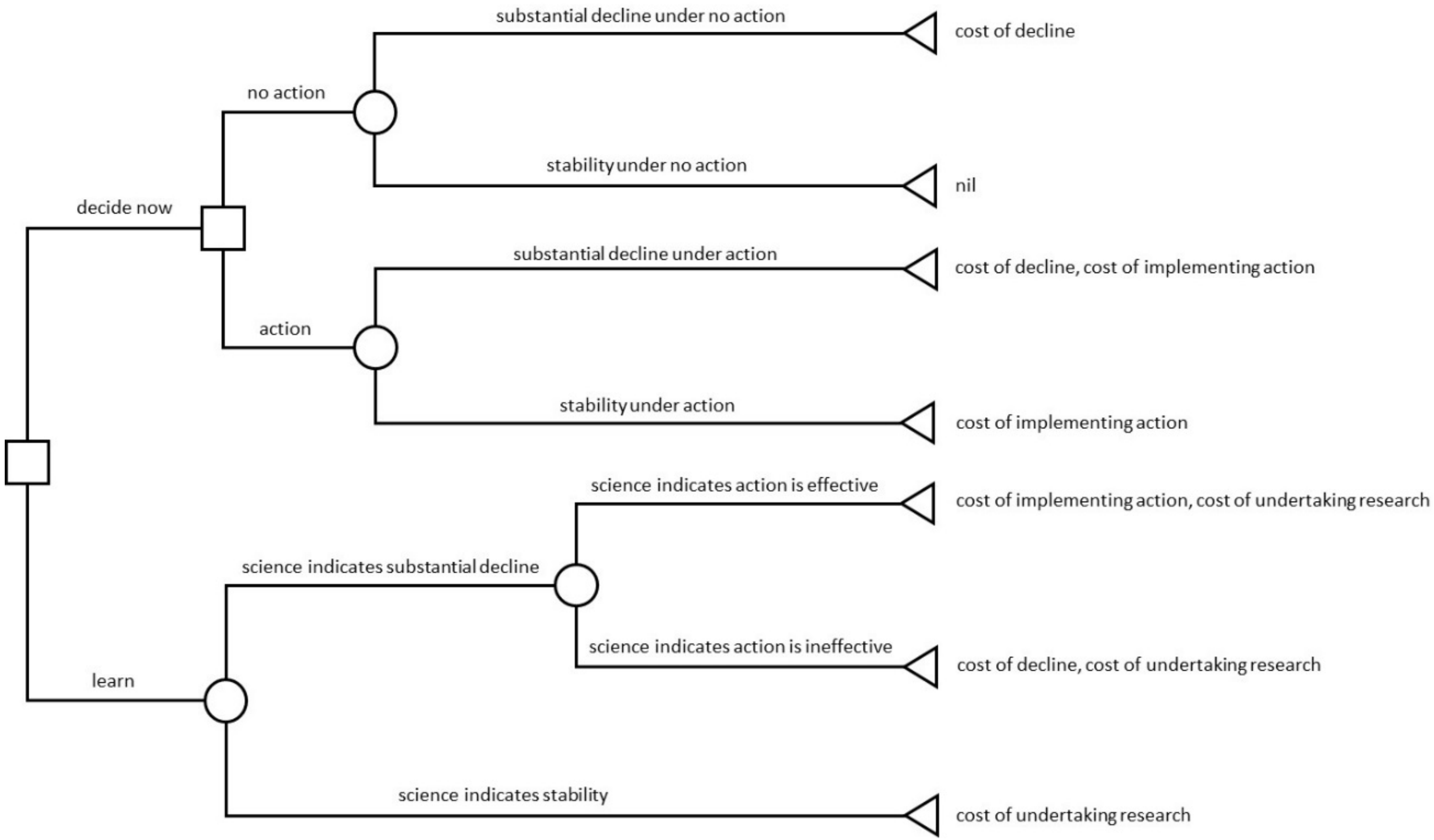
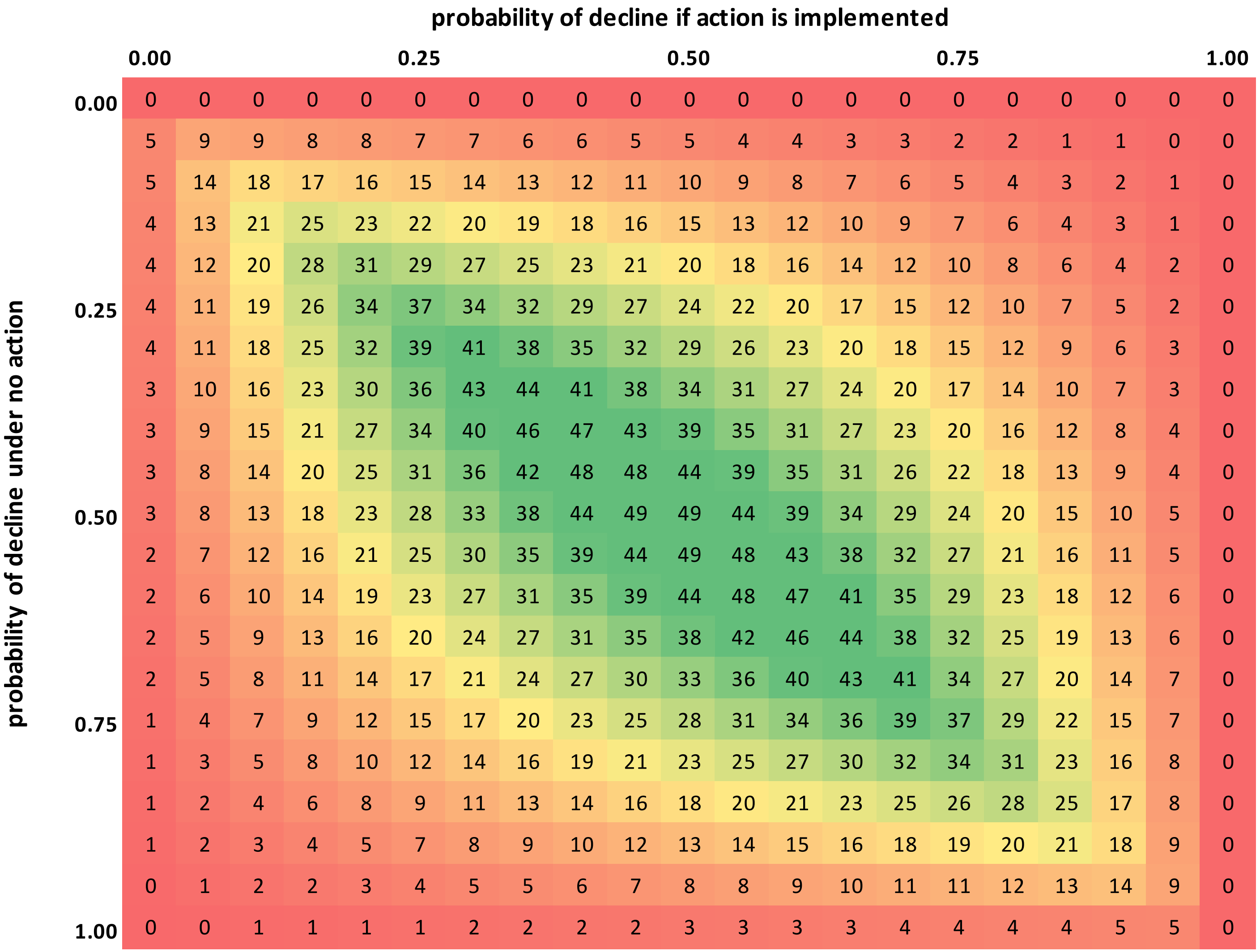


Figure 5. Logic tree for the decision on whether or not to delay a decision on whether or not to act until the knowledge base improves.



	0.00					0.25					0.50					0.75					1.00
0.00	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	5	9	9	8	8	7	7	6	6	5	5	4	4	3	3	2	2	1	1	0	0
	5	14	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1	0
	4	13	21	25	23	22	20	19	18	16	15	13	12	10	9	7	6	4	3	1	0
	4	12	20	28	31	29	27	25	23	21	20	18	16	14	12	10	8	6	4	2	0
0.25	4	11	19	26	34	37	34	32	29	27	24	22	20	17	15	12	10	7	5	2	0
	4	11	18	25	32	39	41	38	35	32	29	26	23	20	18	15	12	9	6	3	0
	3	10	16	23	30	36	43	44	41	38	34	31	27	24	20	17	14	10	7	3	0
	3	9	15	21	27	34	40	46	47	43	39	35	31	27	23	20	16	12	8	4	0
	3	8	14	20	25	31	36	42	48	48	44	39	35	31	26	22	18	13	9	4	0
0.50	3	8	13	18	23	28	33	38	44	49	49	44	39	34	29	24	20	15	10	5	0
	2	7	12	16	21	25	30	35	39	44	49	48	43	38	32	27	21	16	11	5	0
	2	6	10	14	19	23	27	31	35	39	44	48	47	41	35	29	23	18	12	6	0
	2	5	9	13	16	20	24	27	31	35	38	42	46	44	38	32	25	19	13	6	0
	2	5	8	11	14	17	21	24	27	30	33	36	40	43	41	34	27	20	14	7	0
0.75	1	4	7	9	12	15	17	20	23	25	28	31	34	36	39	37	29	22	15	7	0
	1	3	5	8	10	12	14	16	19	21	23	25	27	30	32	34	31	23	16	8	0
	1	2	4	6	8	9	11	13	14	16	18	20	21	23	25	26	28	25	17	8	0
	1	2	3	4	5	7	8	9	10	12	13	14	15	16	18	19	20	21	18	9	0
	0	1	2	2	3	4	5	5	6	7	8	8	9	10	11	11	12	13	14	9	0
1.00	0	0	1	1	1	1	2	2	2	2	3	3	3	3	4	4	4	4	5	5	0

Figure 6. The value of delay and learning the true status of the species or ecosystem and the effectiveness of action under probabilistic judgments of ecological decline in the presence and absence of management action.

Figure 7 shows the corresponding value of learning for the nine scenarios depicted in Figure 4. Perhaps unsurprisingly, it illustrates that delay and learning is a good option only in a subset of circumstances involving high stakes decisions. That is, where the ecological costs of decline and/or the monetary costs of action implementation are high *and* our beliefs in the status and trend of the species or ecosystem or the effectiveness of action are highly uncertain.

Note also that as the costs of implementation rise relative to the costs of ecological decline, the 'zone' in which delay and learning is rational moves toward the bottom left hand corner, where our belief in the probability of decline under no action is relatively high (but not absolute), and our conviction in the effectiveness of action is also reasonably high (but again, not absolute). For many managers, these circumstances may suggest committing to action now as the prudent choice. But the counter-intuitive result reflects the costs of implementing an expensive action that may not be necessary or effective. The corollary is that where the costs of implementation are high relative to the costs of ecological decline, at intermediate or low probabilities of decline and effectiveness, the rational choice is clear: no action and no learning. The decision may be reviewed only where new technology becomes available making action more cost-effective.

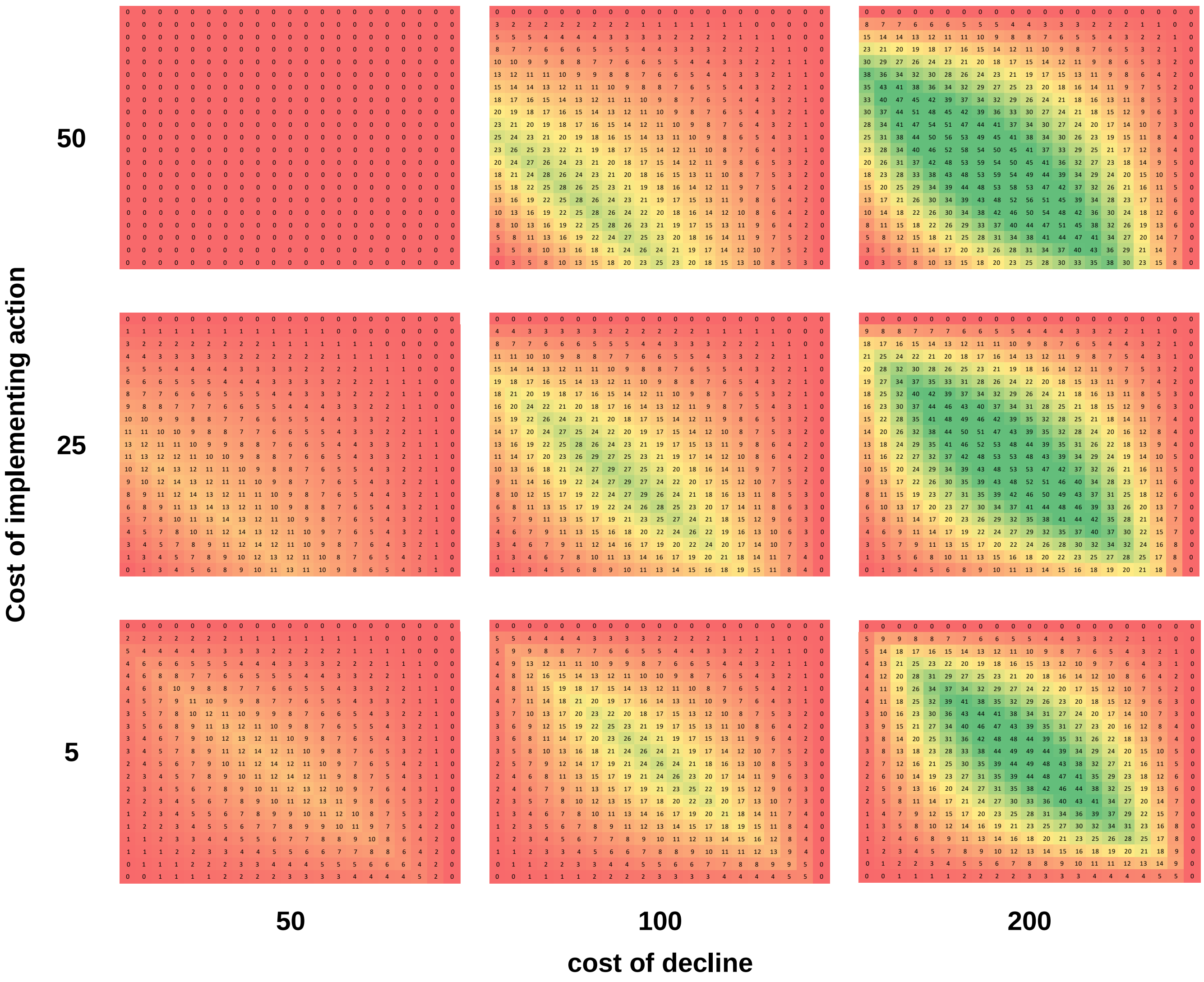


Figure 7. The value of delay and learning for the nine combinations of implementation cost and ecological cost illustrated in Figure 4.

We have used the analysis presented in Figures 2 – 7 primarily to illustrate some core concepts in structured decision-making under uncertainty. But the process of mapping out the logic of the problem, estimating probabilities and consequences, and characterising the value of learning can be used to prioritise investments in science projects. Insights imply that the value of learning via research or monitoring in many circumstances may be modest, and that scarce resources may be better allocated elsewhere. We note that the motivations for knowledge acquisition often extend beyond improving decision-making under uncertainty. Other objectives of learning of relevance to the development of Parks Australia's MERI framework for AMPs include:

- A fundamental need to improve understanding of the values and pressures within AMPs for communication and awareness (as well as an underpinning for evidence-based decision-making).
- An administrative need to report against the full set of AMP values, including use and enjoyment.

The broader resource allocation problem under any MERI framework is a challenging multi-objective problem. Sections2.2.2 and 2.2.3 of this report may assist in its formulation and resolution.

1.3 Resource allocation across program areas

In many settings multiple program areas can influence the extent to which values are effectively protected, conserved or promoted. Here, a common decision problem for organizations is allocating resources across program areas in a way that minimizes risks to values within an overarching resource or budget constraint. For example, some shallow rocky reefs within AMPs are at risk of degradation via colonisation by an invasive algae and/or urchin barrens. The risks may be mitigated, at least in part by:

- Direct control of invasive species, undertaken by *Park protection and management*, and/or
- tighter regulation of fishing effort, undertaken by *Assessments and authorisations,* and/or
- more intense surveillance of fishing effort, undertaken by *Compliance*, and/or
- encouraging behavioural change among stakeholders and users, undertaken by *Communication, education and awareness.*

The listing of specific values and pressures in Parks Australia's management plans (Director of National Parks 2018a,b,c,d,e) provide some of the building blocks for considered allocation of resources. To optimise allocation of resources across the four program areas, we need two additional core elements:

- A model describing the cumulative effect of investment in effort on the value(s) of concern, and
- an estimate of the pay-off of investment in each program area.

Figure 8 uses a fault tree as a simple model for the cumulative effect of investment to illustrate the approach. Fault trees and their accompanying probabilistic logic are described in section 2.1.2 of this report. We note that more sophisticated models may be warranted for high stakes problems. The fault tree shows a (hypothetical) understanding that decline in some natural value (e.g. conservation status of shallow rocky reefs) stems from two pressures, invasive species OR excessive extraction of living resources. Excessive extraction may be an issue if authorisations are too permissive OR there is poor behaviour among authorised users. Poor behaviour arises from insufficient communication AND insufficient compliance. Subject to some budget constraint, the challenge is to estimate the level of investment to be allocated to each of the four program areas.

Now let's say that the pay-off of investment in each program area is estimated using the curves shown in blue in Figure 9. The problem can now be solved using an optimisation algorithm. In this example, we used the readily available Generalized Reduced Gradient algorithm included in Microsoft Excel's solver function (optimisation is described in further detail in section 2.2.2.5). The optimal allocation (the one that minimises risk to the natural value) of a hypothetical $2million budget across the four program areas is shown by the vertical red lines in Figure 9. Note that the approach we outline here can also be used to explore the implications of other budget settings for risk reduction, and to build business cases for additional resourcing.

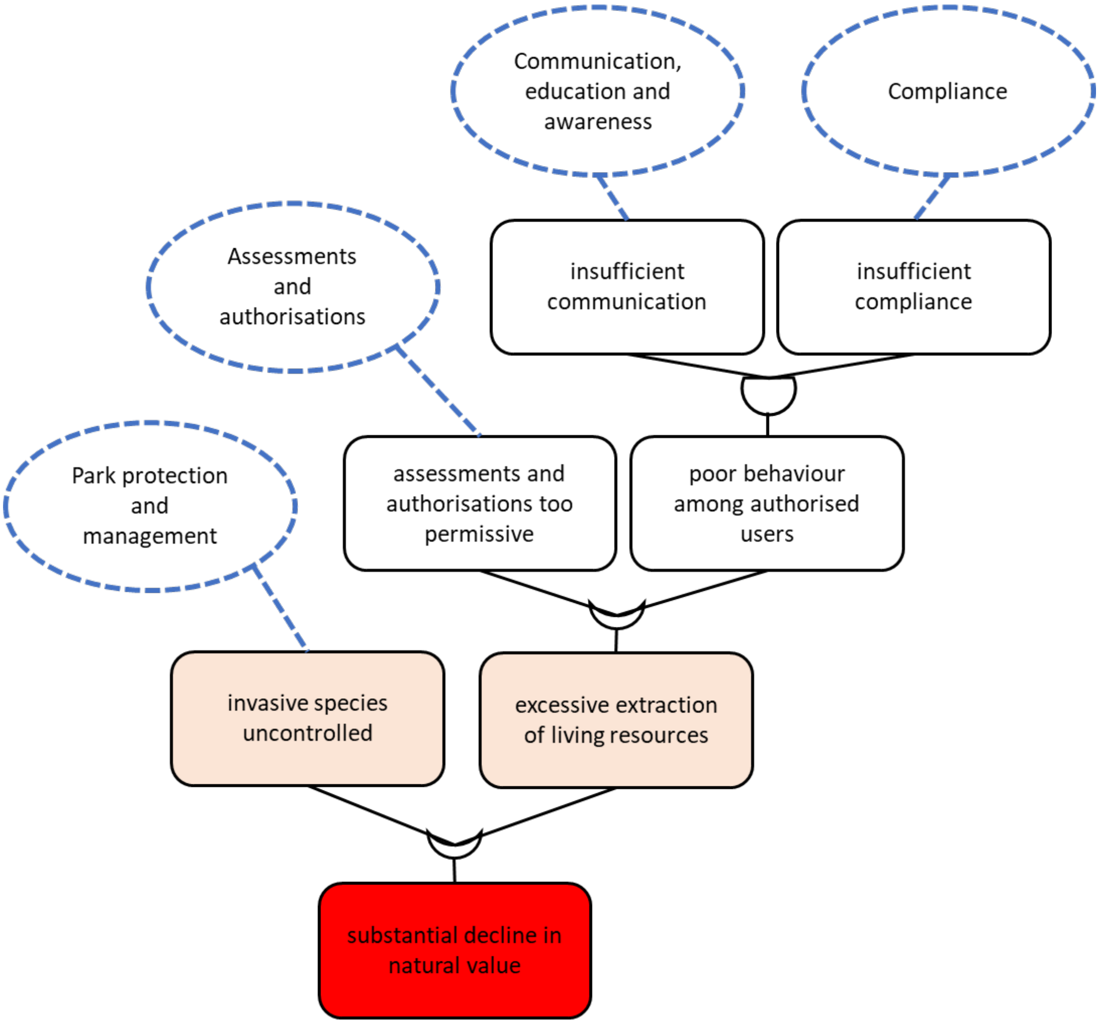
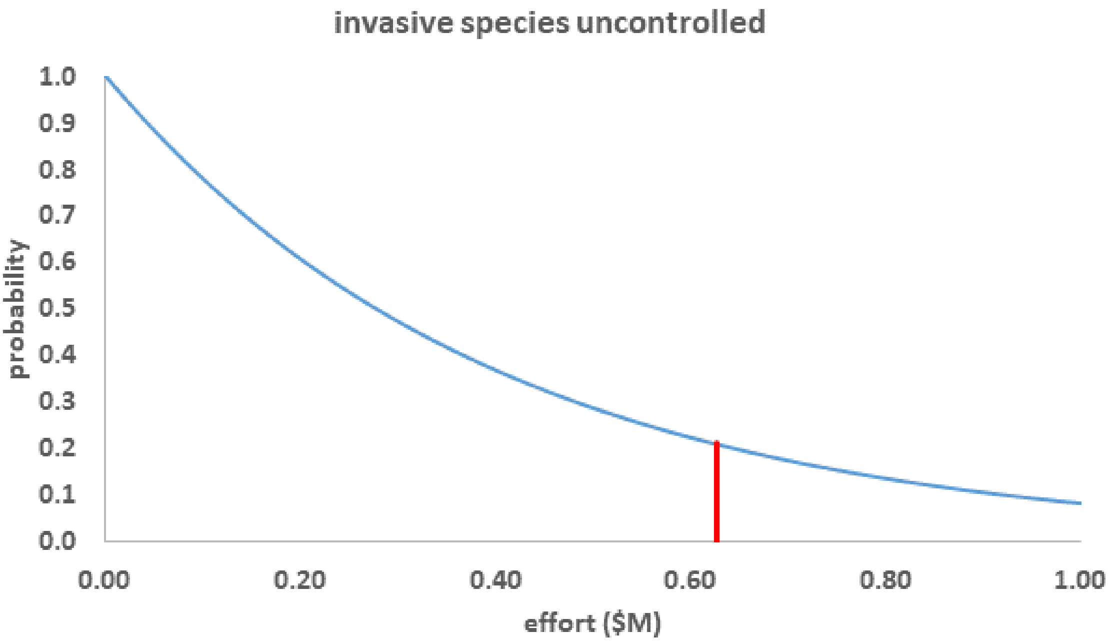


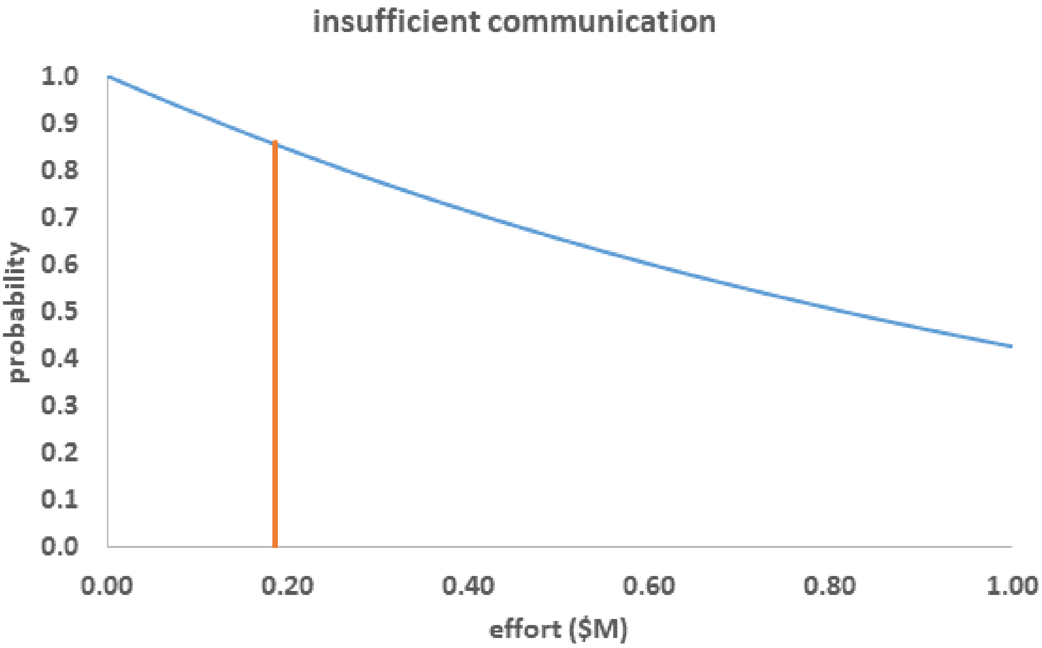
Figure 8. Fault tree depicting qualitative understanding of the individual and cumulative effect of investment among four program areas (blue nodes) in reducing risk to a natural value (red) stemming from exposure to two pressures (orange).

Parks Australia already has reasonably good understanding of some of the natural values occurring within its marine estate and their associated pressures. The most immediate bottleneck preventing the organization from using this approach in routine decisions involving the allocation of resources across AMP program areas is likely to be limited understanding of the shape of the pay-off curves illustrated in Figure 9. Over time, an understanding of the extent to which risks are reduced over a continuum of effort may begin to emerge. In the meantime, the basic approach can be used if the organization is able to make subjective judgments of the approximate pay-off of two or three levels of discrete investment in each program area (see section 2.2.2.1). As information from MERI becomes available, these judgments can be updated and progressively replaced with empirical data.

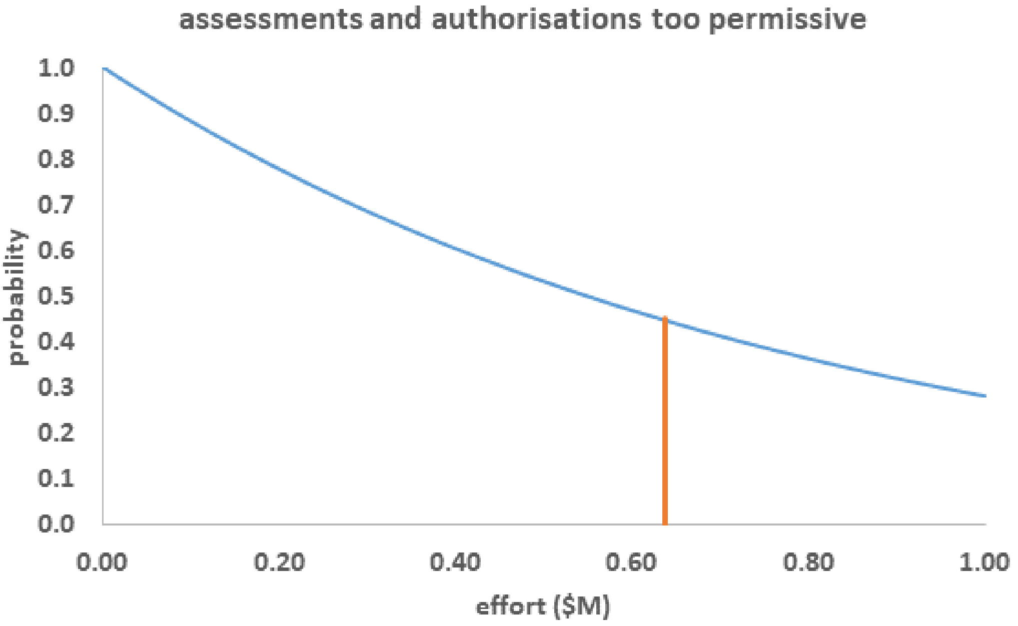
(a)



(b)



(c)



(d)

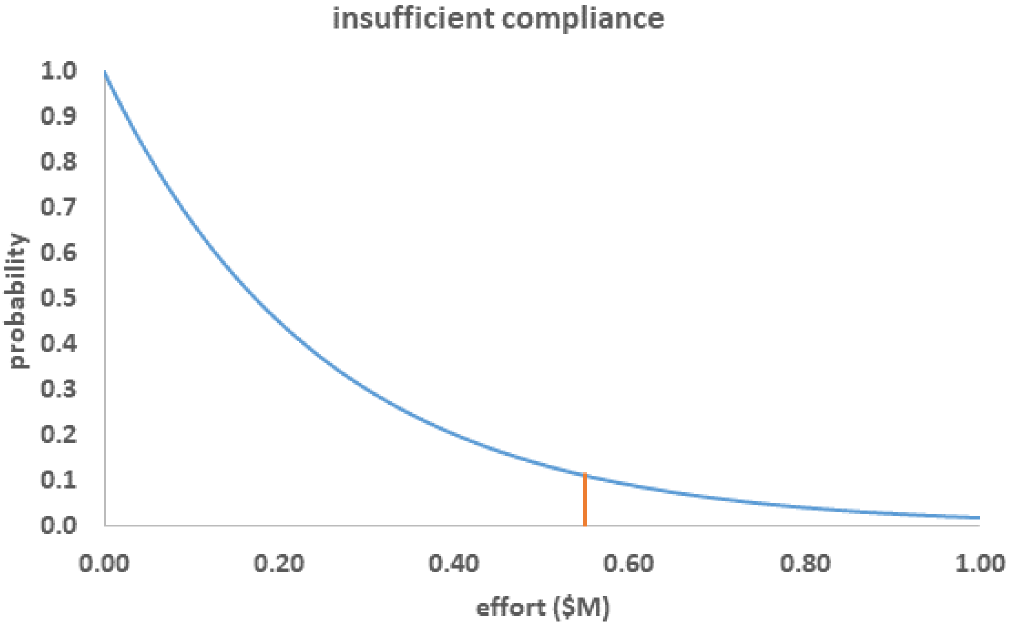


Figure 9. The optimal allocation of resources across four program areas having variable (hypothetical) estimated pay-off curves, under a $2 million budget constraint. (a) Park protection and management, (b) Communication, education and awareness, (c) Assessments and authorizations, and (d) Compliance.

The example problem described here is a single objective resource allocation problem: minimise risk to a single value within some budget constraint. Many resource allocation problems are more complex because they are multi-objective. How can we allocate resources across program areas for all the (conservation and use) values associated with AMPs, or at least the subset of more important values? Multiple objectives inevitably imply trade-offs, or what economists call *preferences*. Sound decision-making often relies on the articulation of preferences as much as probabilistic understanding of cause-and-effect. A prominent illustration of the relevance of preferences in conservation is the distinct tendency to favour charismatic megafauna in the allocation of threatened species management budgets. While this might be consistent with the preferences of some members of society, a subset of scientists and society might consider all species equally important; while others may prefer an allocation of resources that favour the conservation of evolutionary significant species. These views and those of Parks Australia are critical and legitimate elements in the way in which resources are allocated. In section 2.2.1 of this report, we explore the capture of preferences in considerable detail.

2. DECISION SUPPORT TOOLS

A decision support tool (DST) is a platform for integrating, analysing or displaying information to assist decision makers. It may provide insights into the consequences of different approaches to environmental management and conservation, identify the strategy that will optimise a specified objective, identify knowledge gaps, and provide transparency in decision making. There are many examples of DSTs that have been developed by researchers with the intention of assisting environmental managers and policy makers. Here we explore examples most relevant to marine park management.

2.1 Tools to support acceptable risk problems

Assessments, authorisations and permits are acceptable risk problems. Once a threshold of acceptable risk is defined, managers need to consider what approach to use to characterise risk.

Risk analysis encourages decision-making on the basis of *expected* consequences. That is, the calculation of risk as the product of likelihood and consequence is essentially an estimate of expected (dis)utility. While consideration of adverse consequences alone will often suggest the desirability of avoidance or mitigation measures, conditioning estimates of consequence with assessment of likelihood may imply that such measures are not warranted. If estimates of likelihood and consequence are unbiased, then decisions based on risk should lead to greater consistency (Arrow and Lind 1976).In high stakes settings, formal methods in the elicitation of expert judgment can be used to insulate against bias and other cognitive limitations (Hemming *et al.* 2018).

2.1.1 Qualitative risk assessment

Risk assessment has been a common element of planning and management in occupational health and safety, engineering and process industries for several decades, and an International Standard has been developed for its application (ISO 31000:2009). The most commonly deployed approach involves subjective use of a matrix that defines the risk of a hazard as the product of its consequence and likelihood (Table 1). Outcomes depend on the capacity of the analyst to (a) identify ecologically and socially relevant values or assets, (b) elicit an exhaustive list of potential hazards, and (c) use subjective judgment for each potential hazard to estimate the likelihood that an event will occur and the severity of its consequences. The identification of values and hazards may be assisted by a dedicated exercise in horizon scanning (Saritas et al. 2012, Sutherland et al. 2015).

Imposing discrete classes on the continuous concepts of likelihood and consequence can lead to inconsistent assessments of risk. In many instances, the consequences of a hazard may be considered small or catastrophic (or anywhere in between). If an assessor’s mental picture of a hazard involves a small consequence, then the score for likelihood will tend to be high. Catastrophic consequences tend to be associated with a low likelihood score (Figure 10). Variation in interpretation of consequence and likelihood is only problematic where assessors depart from the (idealised) iso-risk curve. But it is reasonable to speculate that such departures will be common given the amorphous cognitive demands of subjective estimation. This arbitrary source of inconsistency can be treated through clearly defining consequence as failure in a specified threshold or target. This approach effectively reduces the method of hazard scoring to the consideration of likelihood alone.

Table 1. The risk matrix suggests use of ordinal descriptors of consequence and likelihood to enable ranking of risks. In the table below, five levels are used to describe the likelihood and consequence of a hazard. Unshaded = low risk, light grey = moderate risk, dark grey = high risk.

Likelihood		**Consequence**				
		Insignificant **(1)**	Minor **(2)**	Moderate **(3)**	Major **(4)**	Catastrophic **(5)**
Almost certain	**(5)**	5	10	15	20	25
Likely	**(4)**	4	8	12	16	20
Moderately likely	**(3)**	3	6	9	12	15
Unlikely	**(2)**	2	4	6	8	10
Rare	**(1)**	1	2	3	4	5

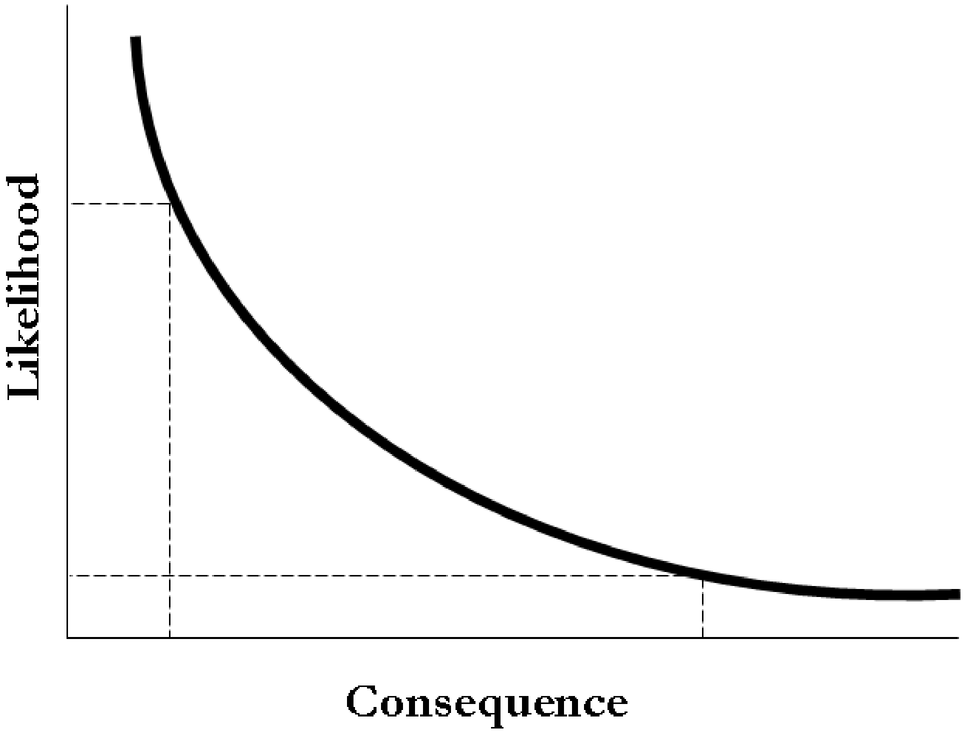


Figure 10. Iso-risk curve (bold curve) for subjective assessment of risk calculated as the product of likelihood and consequence. Individual assessors will view the magnitude of the consequence differently. Large consequences imply low likelihood, and vice- versa (dashed lines). This arbitrary source of inconsistency can be removed through clearly defining consequence as failure in a specified threshold or target.

Although qualitative risk assessment has a number of potential shortcomings, it is important to recognize that a process that encourages the considered identification and assessment of values and hazards is a distinct improvement on *ad-hoc* planning and management. The decoupling of consequence and likelihood insulates against the tendency to overestimate the risk of disastrous but improbable events. Advantages of this ‘minimalist’ approach to risk assessment include:

- It’s simple and fast
- It accounts for probability of harm and magnitude of harm
- It communicates environmental risk in the same language used for financial and social risk
- It provides an informal means of combining data and expert judgment
- It provides an auditable record of priorities

Potential issues associated with a minimalist qualitative approach to risk assessment include the personal and professional biases of the analyst, the ambiguity of language inherent in qualitative assessment, and the distinct tendency for overconfidence in description of the likelihood and consequences of hazards (Burgman 2005). Collectively, these deficiencies can lead to false alarmism (implying a negative impact when none exists) or a false sense of security (implying no negative impact when in fact one exists).

Carey et al. (2007) document extensive use of subjective risk assessment in the identification of management priorities for Parks Victoria’s marine reserve estate. An important insight from their experience was the imperative to treat language-based ambiguities thoroughly before scoring hazards using the risk matrix (Carey and Burgman 2008). By exploring the bases for disagreement through open discussion, this arbitrary source of uncertainty can be largely eliminated, prior to undertaking a risk assessment. However, several iterations of risk scoring and discussion are usually required to satisfactorily deal with language problems. Any residual disagreement will reflect alternative views of cause-and effect associated with a hazard or alternative perspectives on the value of the asset being assessed. Despite its simplicity, Carey et al. (2007) show how the outcomes of a subjective risk assessment can be used to inform priority hazards for management intervention and priority hazards for monitoring or investigation.

Dunstan et al. (2015) recommend a hierarchical approach to risk assessment whereby interactions between pressures (or hazards) and values are hypothesised at level 1, qualitative models are used to formalise understanding at level 2, and interactions are parameterised as quantitative models at level 3. The relative need to progress from a rudimentary level to more sophisticated treatment rests on the estimated magnitude of risk. The risk matrix approach to subjective risk assessment may be useful as a level 1 coarse assessment which may serve to identify priority hazards for detailed treatment under level 2 or level 3.

2.1.2 Quantitative risk assessment - Logic trees

Qualitative estimates are often problematic because of language-based ambiguities (Regan et al. 2002) and because there is a tendency for assessors to confound the task of prediction with value judgments (Hubbard 2009). Quantitative estimates offer much greater clarity. A substantial body of research in psychology and statistical inference illustrates how experts (and lay people) are inclined toward overconfidence in assessing the quality and reliability of their own judgments (Burgman 2005). Elements of poor judgment include:

- insensitivity to sample size - testing ideas on insufficient samples, placing undue confidence in early trends and underestimating the role of uncontrolled, unexpected phenomena or natural variation in producing apparent patterns;
- motivational bias; and
- anchoring judgments on arbitrary or loosely founded estimates made by others.

Burgman (2005) asserts that these frailties interact with the scientific method, leading to irrational inferences and illegitimate appeals to scientific authority. A key source of overconfidence is a failure to appreciate the nature and tenuousness of assumptions upon which judgments are made. Insulation against these ‘frailties’ can be provided, at least in part, through explicit documentation of ideas regarding cause-and-effect, together with explicit description of uncertainty.

Intrinsically, a decision to invest in any management action involves a system understanding that suggests one course of action is preferable to another. Often, these system understandings remain unspoken, unspecified and undocumented. Alternative understandings and courses of action may be entirely plausible, and on consideration, may prove preferable.

Assessment of the consequence and likelihood of hazards requires the risk analyst to form links between cause and effect, which is subject to uncertainties associated with natural environmental variability and lack of knowledge. Experts and non-experts alike are predisposed to overconfidence in their capacity to predict. Lewandowsky and Kirsner (2000) notes that although exceptional performance is a defining attribute of expertise, experts sometimes exhibit striking errors and performance limitations. Hart *et al.* (2005) recommend the use of conceptual models to document assumptions regarding cause and effect and the quantification of these models to explicitly communicate uncertainty in a risk assessment. In marine settings DPSIR framework (drivers, pressures, state, impact and response) has gained considerable traction as the basis for conceptual modelling (Patrício *et al.* 2016).

Quantitative approaches to risk assessment rely on identification of appropriate metrics for characterising consequences (e.g. quality adjusted recreation days, catch per unit effort). The core business of conservation management is to extend the persistence and functionality of species and ecosystems for as long as possible. On geological time scales, all species go extinct and all ecosystems eventually collapse, even in the absence of anthropogenic agents. Recognising the additional risk posed by human pressures, conservation agencies seek to prolong persistence over socially relevant time horizons. One mechanism for doing so is the regulatory approach underpinning acceptable risk decisions and one metric for characterising acceptable risk is estimated time to extinction, or expected extant years.

The IUCN assessment protocols for assessing threat category for species imply extensive time horizons. The lowest threat category, ‘vulnerable’, equates to a 10% chance of extinction over 100 years. We might say a species is non-threatened if we expect time to extinction to extend beyond say 1,000 years. Let’s say that Parks Australia undertakes the substantial task of specifying the extent to which it seeks to maintain viability for biota in different protected area categories and parks. For example, in some hypothetical park, we may say for illustrative purposes that management seeks to maintain species occurring within IUCN Category IV for 800 years. For Category VI, where sustainable use is accommodated, we might set a lower threshold of say 400 years.

In all circumstance, management will be uncertain whether a species is threatened or non-threatened. For simplicity, let's say that a species is threatened if it's expected to persist for just 100 years; and if it's non-threatened, it is expected to remain extant for 1,000 years. Imagine the *Authorisations and Assessments* unit receive a proposal for an activity that might adversely affect a species within IUCN Category IV. In the absence of the proposal going ahead, the unit estimates the probability of the species being threatened or non-threatened:

State	**probability**	**expected extant years (EEY)**
threatened	0.20	100
non-threatened	0.80	1 000

If the proposal goes ahead, we estimate an increased probability of threat:

State	**probability**	**expected extant years (EEY)**
threatened	0.30	100
non-threatened	0.70	1 000

The information above can be communicated more clearly in a logic tree (Figure 11).

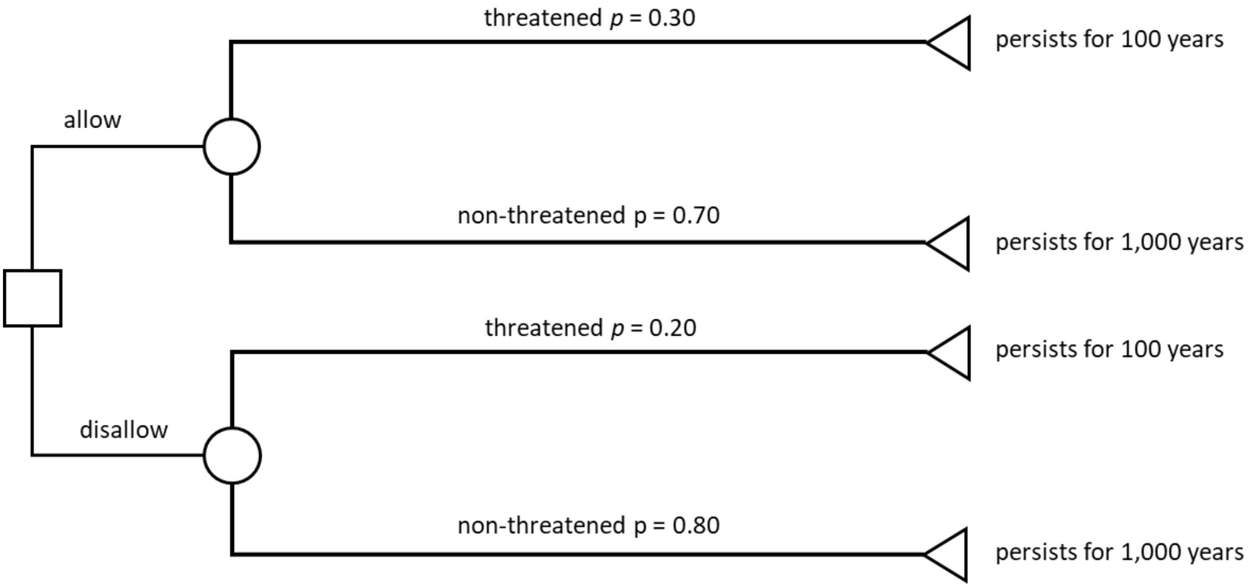


Figure 11. Decision tree showing the decision node (square) to allow or disallow a proposal that has uncertain (circles) consequences.

Should management allow the proposal? Without the proposal, the species satisfies the acceptable risk threshold of 800 years. The probability-weighted expected extant years (EEY) is 0.20 × 100 + 0.80 × 1 000 = 820 years. If the proposal is allowed, we estimate EEY to be 0.30 × 100 + 0.70 × 1 000 = 730 years. The proposal is disallowed because it does not satisfy our acceptable risk threshold.

A readily accessible tool for documenting ideas of cause and effect that might lead to goal failure (or failure in satisfying an acceptable risk threshold) is a fault tree. Fault trees are a form of logic tree. They are an extension of conceptual models that accommodate quantification and uncertainty (Burgman 2005). They use some standard symbols, including (Hayes 2002):

○	Basic event: events that indicate the limit of resolution of the fault tree.
◇	Underdeveloped event: indicating the level of detail could be greater.
∩	AND gate: output occurs only if all inputs are true (or occur simultaneously).
⋀	OR gate: output occurs if any input is true.
□	Event: an event or condition within a fault tree.

If an expert or manager can estimate the likelihood of observing basic events in a fault tree, the various AND and OR statements that make up the tree can be subjected to ordinary probabilistic calculus to estimate the likelihood of all events. Events may be mutually exclusive or independent. If they are mutually exclusive, then the probability that one or the other will occur is given by

$$p(A \cup B) = p(A) + p(B)$$

and the chance that both will occur is, by definition

$$p(A \cap B) = 0$$

If two events are independent, then the chance that either one or the other, or both, will occur, is

$$p(A \cup B) = p(A) + p(B) - p(A \cap B)$$

where

$$p(A \cap B) = p(A)p(B) \text{.}$$

For three events, A, B, and C, the chance of (A, B or C) is given by

$$p(A \cup B \cup C) = p(A) + p(B) + p(C) - p(A \cap B) - p(A \cap C) - p(B \cap C) + p(A \cap B \cap C) \text{.}$$

Calculation of the probability of all events in a tree relies on estimates of the probability of basic events, which themselves are subject to uncertainty. To better describe this uncertainty, upper and lower bounds for basic events can be included and the probabilistic calculus described above can be applied to intervals rather than point estimates (Neumaier 1990).

The outcomes of a risk analysis can inform where investment in management action is warranted and where it is of low priority. Where uncertainty is high, it can also identify knowledge gaps that might require research or monitoring.

The most widespread and prominent use of AND/OR logic in quantitative assessment of conservation is the rule set developed by the IUCN for the assessment of conservation status for species (IUCN 2001). Figure 12 shows the logic tree embedded in the IUCN protocol. Recently, this approach has been extended to the assessment of entire ecosystems (Keith et al. 2013), with assessments to date including the marine examples of the Aral Sea (status = collapsed) and Caribbean coral reefs (endangered).

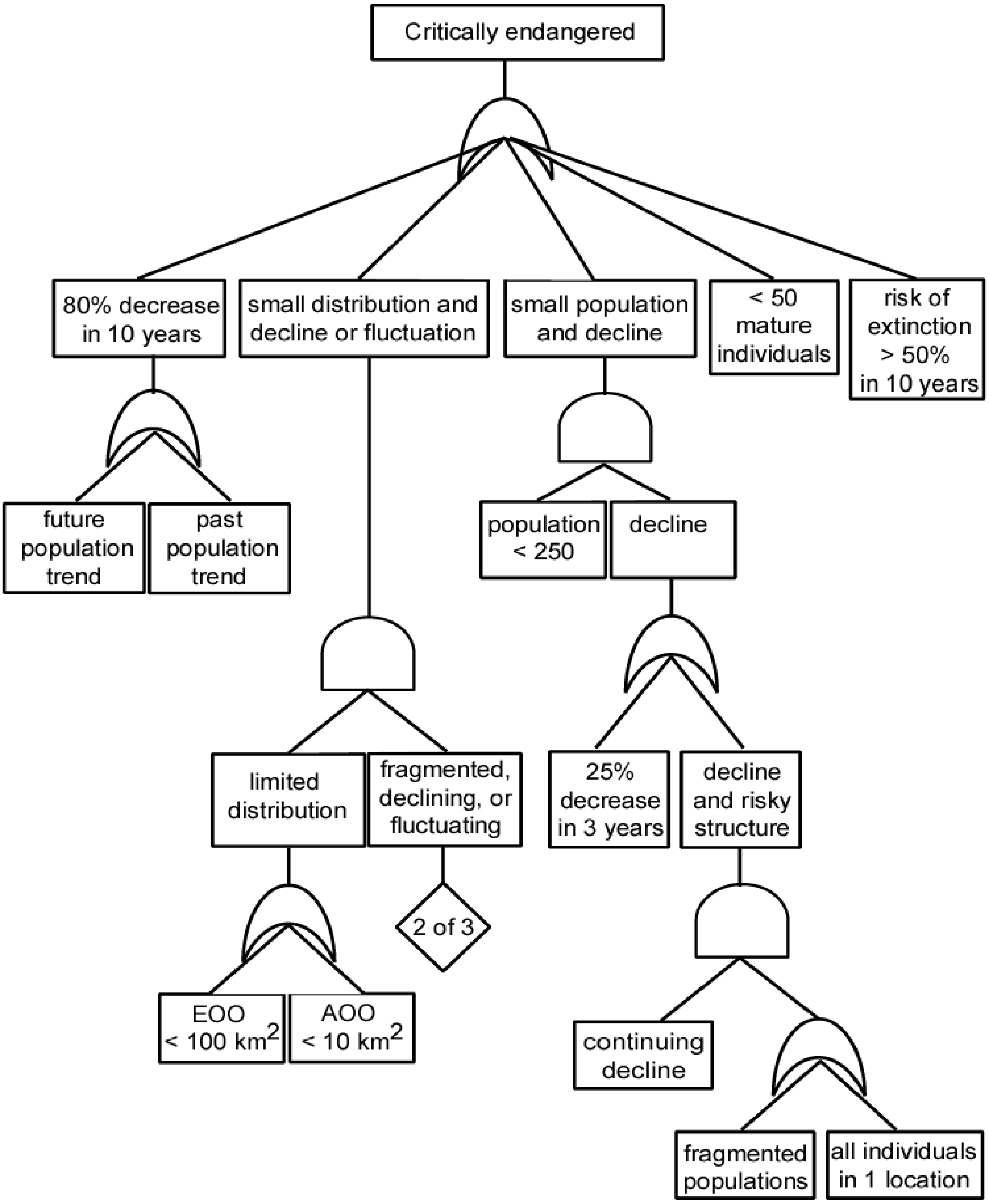


Figure 12. The logic tree underpinning the IUCN's protocol for assessment of a species' conservation status as being critically endangered or not critically endangered. Source: Burgman (2005).

2.1.3 Quantitative risk assessment - Bayesian Belief Networks

Bayesian Belief Networks (BBN) can also be regarded as a tool that builds on insights from conceptual models. BBN’s consist of a graphical structure and a probabilistic description of the relationships among variables of a system. The graphical component is akin to a conceptual model of cause and effect, where system variables are represented as nodes, and arcs between nodes imply that the state of a ‘child’ variable is in some way dependent on the state of one or more ‘parent’ variables. A BBN allows complex causal chains linking actions to outcomes to be factored into an articulated series of conditional relationships (Borsuk *et al.* 2004). The capacity of BBN’s to incorporate empirical observations, system sub-models, and expert opinion makes their application in complex systems appealing. Their basis in Bayesian inference also means BBN’s can be readily updated as new information from research and monitoring becomes available.

Advantages of BBN’s include (Hart *et al.* 2005):

- BBN’s are useful when scant data is available
- They can synthesize scientific data, existing models and expert opinion
- They can be used to formalize understanding
- They can identify and prioritize important variables (sensitivity analysis)
- They can be used to explore the effect of different management actions (predictive)
- They can be updated easily in the light of new information
- They provide a probability estimate for the likelihood of complying with endpoints identified in an Ecological Risk Assessment.

Bayesian network models provide a probability of an outcome rather than a discrete (deterministic) outcome. From the probability distribution, a mean (expected) outcome and confidence interval can be determined. The components of the BBN are linked through a series of conditional probability tables (CPTs). These provide a probability of a particular outcome (or outcome level) being achieved given the observed combinations of inputs. A further advantage of a BBN is that the inputs can also be given a probability distribution. That is, when there is uncertainty as to which category the input falls into (for example, whether the scale of an offset is sufficient or not) then probabilities can be (subjectively) assigned to different levels, and these can be used to determine the most likely outcome under uncertainty.

Bayesian networks have been applied to a number of environmental and natural resource management applications, particularly when the effects of qualitative as well as quantitative factors are of interest. Examples include fisheries (e.g. Little *et al.* 2004; Pollino *et al.* 2007; Cole 2010; Martin-Ortega *et al.* 2011; Pascoe *et al.* 2011a; van Putten *et al.* 2013), ecosystem services (Landuyt *et al.* 2013; Sun and Müller 2013), assessment of marine offsets (Jennings et al. 2015) and feral animal management (Lethbridge and Harper 2013).

An example of the structure of a BBN, in this case used to develop a social, economic and cultural report card for a coastal industrial harbour (Pascoe et al. 2014c), is given in Figure 13. Each level of the BBN is linked through a CPT that describes the probability of an outcome (in the child node) given the observed conditions in the contributing parent node. CPTs can be derived though observed outcomes and inputs, and/or expert opinion. Software packages for building BBNs are readily available and include *Winbugs*, *Netica, Genie* and *Hugin*.

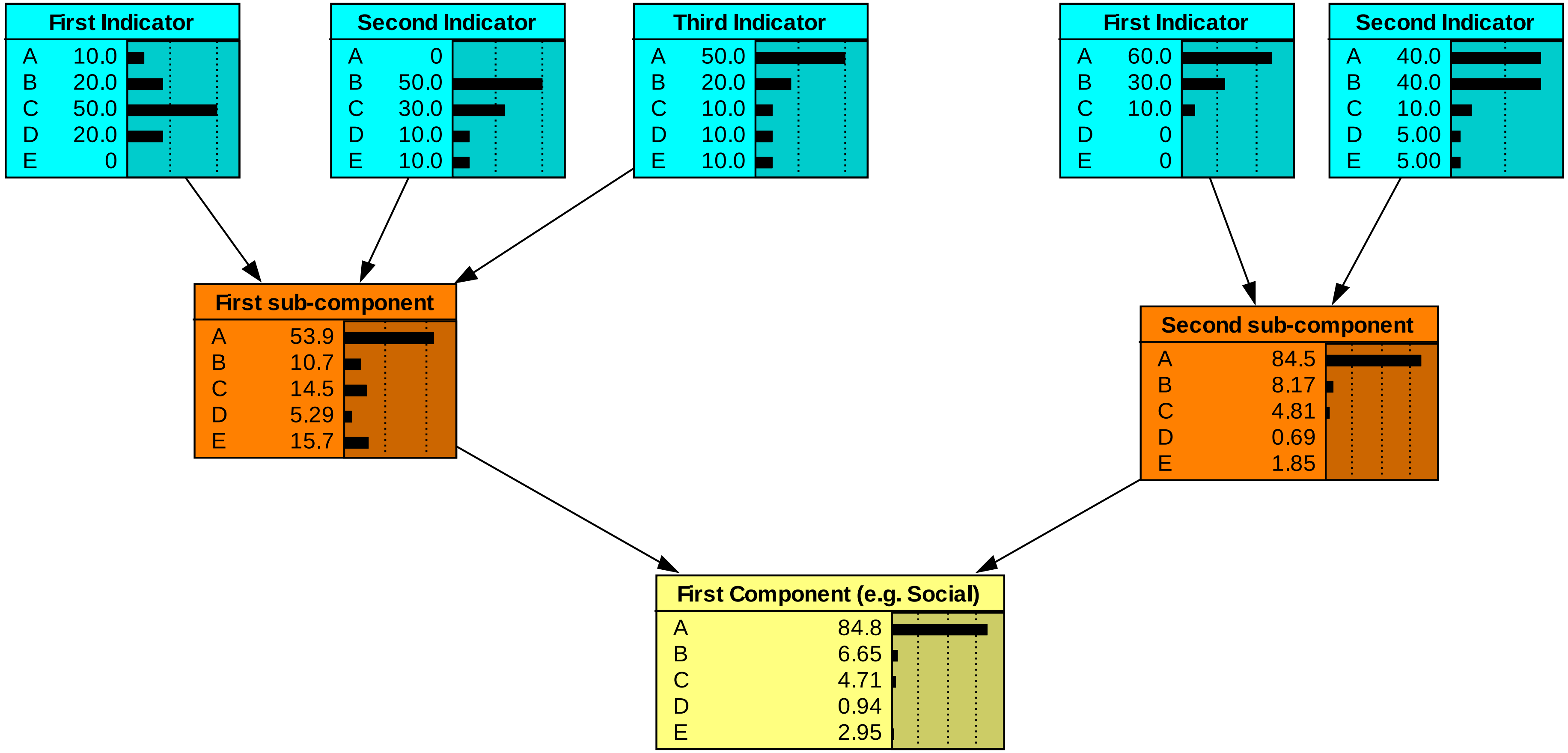


Figure 13. Example structure of a BBN. The numbers in each box represent the probability that the level of each indicator or variable (in this case A-E) is realised.

2.1.4 Quantitative risk assessment –Monte Carlo simulation

Stochastic simulation models offer the highest resolution approach to risk assessment available. Computational advances mean that models may explicitly incorporate temporal dynamics, spatial variability, parameter, shape and structural uncertainty, together with uncertainty arising from environmental variation (Burgman 2005).Use of such models is well established in fisheries management, including applications involving multiple target species and ecosystem-based approaches (e.g. Fulton et al. 2014, Plagányi et al. 2014). Similar approaches are used in conservation in terrestrial settings (e.g. Lindenmayer and McCarthy 2006).

The cost of resolution is a high demand for data or credible expert judgment in specifying and parameterising models. Dunstan *et al* (2015) suggest these costs may be warranted for key values that are exposed to high risk pressures, demanding a more diligent approach to risk assessment.

2.2 Resource allocation problems

Resource allocation problems are often multi-objective problems. There are three core elements to any multi-objective resource allocation decision problem:

- Alternative allocations of resources,
- expected consequences against two or more objectives (e.g. conservation status and cost), and
- trade-offs underpinned by an understanding of preferences (e.g. preference for saving a species versus preference for cost savings).

Before exploring tools and techniques that assist in the selection of better alternatives among candidates, we review approaches to the capture of managers' (and/or stakeholders') preferences.

2.2.1 Preferences - valuation of market and non-market consequences

Economic theory suggests that preferences are reflected in individuals' willingness to pay for particular environmental outcomes. However, as not all environmental assets or outcomes have an explicit market price, non-market values need to be estimated. The economic concept of value is based on the premise that the purpose of economic activity is to increase the well-being of individuals who make up the society and that these individuals are the best judge of their wellbeing. It is, therefore, strongly anthropocentric or human-focussed, in its orientation. It is further assumed that individuals' preferences are expressed through the choices and (implicit) trade-offs they make (or are willing to make) between different goods and services, given constraints such as their incomes and the other resources available to them. In most societies, these choices are most regularly expressed through people's behaviour in markets, where buyers express their willingness to pay (make trade-offs) through their demand for goods; and producers reveal information about the economic costs of making goods available through their supply behaviour. Within this context, the conceptually correct measure of the value people place on changes in the quantity or quality of particular goods and services is the change in the net benefits they experience from participating in the market; that is by measuring changes in consumers' and producers' surplus, where surplus refers to a net benefit. In the case of producer surplus, this represents the difference between the full cost of production and the price received, and is observed in the market as a measure of economic profit. In contrast, consumer surplus represents the difference between what consumers are willing to pay for a good, and the price that they are required to pay. These measures further imply the use of a common monetary metric for the measurement of economic values.

That said, economists emphasise the wide range of types of values that can affect individual's wellbeing through the concept of Total Economic Value (TEV), some of which may not, and many of which cannot, be exchanged in markets. Hassall & Associates (2004) provide an overview of the potential TEV of marine resources. At the highest level, TEV recognises that people benefit from goods and services both through their use and non-use (see Figure 14). These categories are further disaggregated to highlight the potential for both direct and indirect use; and various categories of non-use, such as bequest values (e.g. the value to current generations of ensuring the environmental asset is available for future generations) and existence value (i.e. the value of knowing an environmental asset continues to exist even if you never experience it directly).

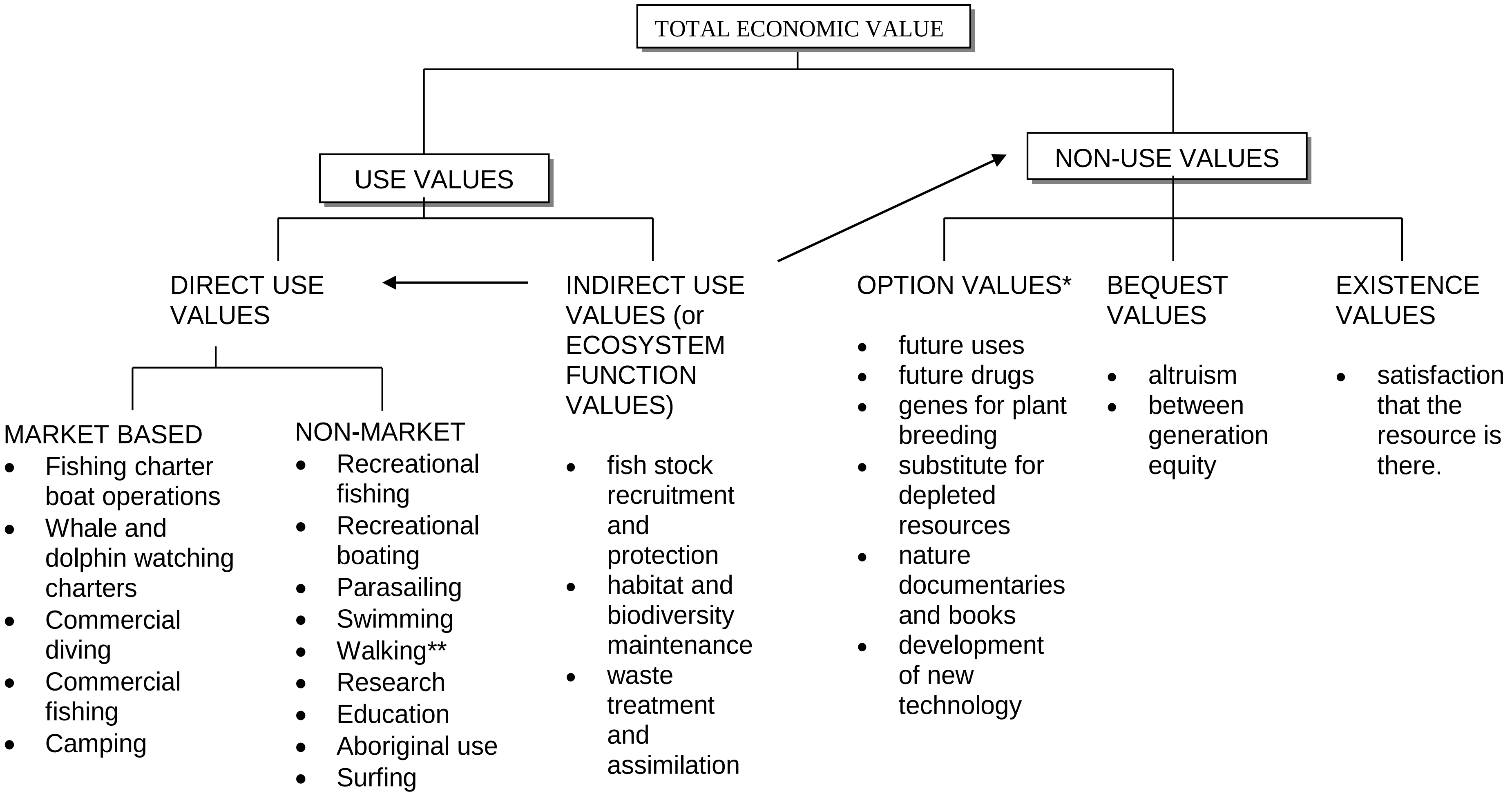


Figure 14. Total Economic Value of marine resources (Source: Hassall & Associates 2004)

The importance of adopting a comprehensive approach to valuation is well accepted, and a number of approaches have been developed to allow the valuation of TEV. While producer surplus can be identified for all goods traded in markets, consumer surplus can be identified for both marketed and non-marketed goods, suggesting the need for valuation techniques that lie beyond the analysis of actual markets for the goods and services being traded.

As a conceptual framework to guide applied economic valuation, TEV is closely linked to the emerging concept of ecosystem services, which now dominates the ecosystem assessment processes and policy agendas in many countries (Jax et al. 2013). The central idea of ecosystem services is that ecosystems contribute to human well-being and many authors show the link between components and processes of the marine environment; intermediate ecosystem services, final ecosystem services and the goods and benefits (see Potts et al.2014) that are ultimately of anthropogenic instrumental value, and hence the object of economic valuation. Jax et al. (2013) caution against tying the newly emerged concept of ecosystem services to any single value dimension or valuation framework.

While many conceptual and practical issues remain to be resolved with economic valuation methods (see for example Pendletonet al. (2007)), the field is relatively well developed and there are a large number of peer-reviewed articles and books and grey literature reports providing technical guidance (e.g. Haab and McConnell (2002) and Baker and Ruting (2014)) including empirical examples by method (e.g. Wallmo and Edwards 2008; Kragt and Bennett 2011; Pascoe et al. 2014b), location (Hassall& Associates Pty Ltd 2001) and type of ecosystem service (e.g. Marreet al. 2015).Here we provide only a brief overview of the methods available to value the market and non-market consequences of environmental management, focussing on examples of their use in informing decision-making in the marine context, and where possible in Australia.

Market values approaches

As noted above, within the economic valuation framework, markets are viewed as playing a pivotal role as value-articulation institutions, through which individuals express preferences and reveal trade-offs, thereby providing a sound basis for economic valuation. It is little wonder then that where possible, economists have turned first to actual market data for information about the values of ecosystem services. The two main market valuation approaches are: direct market value and cost-based methods.

The first of two direct market methods is known as the *price-based approach*. As suggested above, for ecosystem goods and services - such as fish for human consumption or for conversion to fishmeal, or for marine-based, commercial ecotourism experiences – if there are sufficient observations of market trades, it is possible to use standard econometric techniques to estimate values for both buyers and sellers. Furthermore, under certain conditions and in ideally functioning markets, market prices can provide a measure of the social value of incremental or marginal units of a good or service.

As new markets emerge for a greater range of ecosystem services, the scope for such valuation may increase, although the robustness of the estimates derived from this approach depend on the assumption of well-functioning markets, or practitioners' ability to adjust observed market data to reflect market failure or other distortions.

A second direct market value approach is the *production function-based* approach which is used to estimate the value of goods and services that contribute to the production of a marketable good. The approach is relevant in cases where the biological resource or ecosystem service to be valued is an input into the production of a marketed good. For instance, a marine habitat or feature (e.g. seamounts), is an input into the production of fish. The biophysical relationship between inputs and outputs in the production process (known as the production function) can be used to infer values for the inputs even when they are not themselves marketed. The 'derived demand' for the ecosystem service as an input can be estimated from the demand for the final marketed output in association with information from the biophysical production function. The approach requires good information on both the biophysical production function (i.e. the way in which habitat interacts with other inputs to produce fish) and information on the market for the output, fish. Such methods are essentially supply-side approaches to valuation relying on estimates of consumer and producer surplus in downstream markets (Hoaglandet al. 2013). The estimates provided by these methods represent a lower-bound estimate of value for the ecosystem asset being valued (e.g. seagrass) as such assets may provide use values to other user groups and also non-use values.

McArthur and Borland (2006) use measures of the relative association between species and habitat and estimates of the value of the marginal productivity of habitat to estimate the impact of seagrass decline on the secondary production of seven commercial fish species in South Australia. They then estimate the value of a 16% decline in seagrass as the foregone market value (rather than net benefit) of this decline in production. Using an analytical approach, Sanchirico and Mumby (2008) develop and interrogate a bioeconomic model incorporating multiple habitat-fishery linkages to determine the marginal opportunity cost (foregone economic rental value of the fishery) from a loss in mangrove habitat. *Cost-based approaches* include methods that involve observing the purchase behaviour of people when ecosystem services are threatened. Values are thus inferred from the amount of money people are willing to spend to avoid or mitigate the consequences of ecosystem service loss, and are known as preventative expenditure and averting behaviour approaches. For example, if a particular species is under threat of extinction, the cost of a breeding program or of installing devices on fishing boats to prevent its accidental capture may be used to estimate the benefit being provided by its continued survival. A second cost-based approach involves the estimation of how much it would cost to replace the lost ecosystem service benefit with a substitute. The cost of replacing natural with artificial reefs and of stock enhancement programs to maintain fish stocks in the face of degraded or lost habitat can be used to infer a lower bound on the value of the ecosystem service to society.

While the dominant approach to estimating the value of regulating ecosystem services (deGroot et al. 2012), cost-based approaches do not provide conceptually correct valuation methods based on measures of changes in economic welfare. However, subject to ensuring that replaced ecosystem services are identical to those that are lost; and that the decisions to invest in such measures are taken giving due consideration to the lost benefits, such methods can provide an inexpensive first approximation of value.

Revealed preference approaches

Revealed preference (RP) methods look to the observed choices that people make in actual markets that are related to the ecosystem service that is to be valued. For example, while recreational fishers do not purchase the fishing experience *per se*, they do make purchases related to the activity in the markets for fishing gear. Likewise, the amenity value of coastal views and access are purchased indirectly in the surrogate market for coastal properties. The two main revealed preference methods are the travel cost method and the hedonic method.

The *travel cost (TC) method* is based on the premise that, while much recreational activity is not exchanged in markets, individuals engaging in recreational experiences do incur costs, in the form of direct expenses (on say gear, bait, boats and fuel) and opportunity costs of time taken to travel to a recreation sites (such as a lake or a beach). Such costs are assumed in TC to form the implicit price of visiting the site, and the demand for the site (i.e. the number of visits to the site in a given time period) is then estimated as a function of this price and the socioeconomic characteristics of the individual. Consumer surplus associated with the use value only of the site can then be derived from this estimated demand relationship. Alternatives to the classical, single site TC model have been proposed whereby recreational site choice is cast as a discrete choice problem. Visitors are assumed then to select sites based on their preferences for site characteristics (say catch rates and crowding) and cost.

The use of the TC method for estimating the non-market use value of environmental amenities is well established (Mendelsohn and Olmstead, 2009), although applications are less common in the marine than in the terrestrial environment. These include studies of dive-related tourism (Seenprachawong 2003; Ahmed *et al.* 2007; Brander *et al.* 2007; Tapsuwan and Asafu-Adjaye 2008; Pascoe et al. 2014b), biodiversity (Schuhmann *et al.* 2013), recreational fishing (Shrestha *et al.* 2002; Rolfe and Prayaga 2007; Prayaga *et al.* 2010; Pascoe *et al.* 2014a) and valuation of marine parks (Chae et al. 2012; Mwebaze and MacLeod 2013). In Australia, a number of TC studies have estimated the value of changes in the quantity and/or quality of recreational fishing opportunities. For instance, Ward et al. (2012) estimate the net use value of game fishing trips to Bermagui and Port Stephens; Ezzy et al. (2012) determine the recreational value of the non-commercial southern bluefin tuna catch at Portland, in southwest Victoria, emphasising the importance of this group of users in the management of the overall fishery.

The TC method has been used more specifically to estimate economic use values for a range of recreational activities associated with marine parks. For instance, Chae et al. (2012) use a TC model to estimate the value of marine nature-based tourism in a Marine Nature Reserve in the UK; Mwebaze and MacLeod (2013) estimate the value of tourist visits to marine parks in the Seychelles; and Pascoe et al. (2014b) focus on the value of dive tourism in marine parks in three countries in South East Asia, and use their TC model estimate of the price elasticity of demand to predict the effect on visitation of the introduction of an entry fee intended to offset the costs of managing marine parks. In a similar Australian study, Farret al. (2011)estimate the welfare loss to live-aboard dive recreationalists in the Northern Great Barrier Reef of the Environmental Management Charge. Rolfe and Gregg (2012) identify local recreational use values for recreation in and adjacent to the Great Barrier Reef Marine Park, focusing on beach, island, and fishing, boating and sailing trips; and Gillespie Economics (2007) uses TC to measure recreation use values for domestic visitors to four state marine parks in NSW.

Combining standard TC data with survey-based information about contingent behaviour (i.e. how recreationalists' visitation rates might change if there were a change in the quality or quantity of the recreation asset) enabled Rolfe and Gregg (2012) to estimate beach recreation values for 1400 kilometres of the Queensland coast and to predict the marginal effects on such values of potential declines in water quality. In a similar study, Rolfe and Dyack (2011) estimate the change in consumer surplus of recreationalists that would result from altered access to the Coorong. Likewise, Kragt et al. (2009) use a questionnaire to elicit data on both travel costs and contingent behaviour to calculate the welfare loss to dive and snorkelling tourists on the Great Barrier Reef from reef degradation.

The *hedonic price method* (HP) is another method that uses market behaviour to infer preferences for non-marketed goods and services. It is based on consumer theory which recognises that the utility of a good is related to the attributes of the good, and is operationalised by using data on the observed prices and attribute content of multi-attribute goods and services to deconstruct total price to reveal the marginal implicit prices of individual attributes. The HP method is most often used to value environmental amenities or environmental quality that affect the price of residential properties, and is limited to those that can reasonably be assumed to directly affect consumer behaviour and market values. For example, because each coastal property represents a unique combination of attributes, the price the consumer is willing to pay for a property will be determined by a range of attributes and the degree to which these attributes are provided by the property. This approach derives the value of specific attributes of a property by undertaking regression analysis which differentiates property values based on a range of attributes including, for example, physical characteristics, accessibility to services, neighbourhood and environmental amenity characteristics such as view, water frontage and access to beach.

There are few examples of HP methods being applied to value non-market attributes of the marine environment. van Beukering et al. (2009) use the HP method to estimate the contribution of reefs to the amenity value of real estate in Bermuda. In Australia, Hodgkinson and Valadkhani (2009) used a commercially available database to estimate the value residents were willing to pay to live in close proximity to Lake Illawarra, coastal NSW, where the foreshore amenities were upgraded and the degraded water quality was undergoing substantial restoration.

While the main advantage of revealed preference methods is their reliance on actual market data, market imperfections can distort observed prices and hence estimated ecosystem service values. The robustness of such estimates, which can involve complex statistical analysis, therefore depends on the availability of large high-quality data sets and the validity of the assumed relationship between the ecosystem good or service being valued. Revealed preference methods cannot be used to estimate non-use values.

Stated preference approaches

The above methods provide the means to estimate the use (direct and indirect) values for ecosystem services but are unable to provide decision-makers with information on the values held by those who do not directly use such services, or actively benefit from them. For example, members of a community may be willing to pay for the non-use values of marine ecosystem services; and to support the provision of use values (such as recreation) that they do not directly benefit from. Stated preference approaches to economic valuation have emerged to account for such values, the primary methods being choice experiments and contingent valuation. Both methods rely on using surveys in which respondents are presented with hypothetical scenarios regarding the provision of ecosystem services and payment options, and are asked to state their preferences.

Central to the *choice experiment (CE)* method is a stated preference survey in which respondents are asked to make choices between different (hypothetical) alternatives presented to them. CE surveys typically describe several hypothetical scenarios that will lead to different outcomes, where each outcome is described by the levels of different attributes, including a monetary attribute (or cost), which together describe the good to be valued. Survey respondents are presented with a series of questions, each including two or more alternatives (choice sets) from which they are asked to choose their preferred option.

Preferences are then inferred from the stated choices that people make between the constructed (hypothetical) combinations of attributes.

The theoretical foundations of choice experiments lie in Lancaster's theory of value (see HP above) and on random utility theory (McFadden 1986). In essence, the latter posits that the utility an individual derives from a particular choice is a latent variable that is observed indirectly though the choices people make. Utility is further assumed to comprise an observed or systematic component and a random unobserved error component. In deriving people's preferences from survey data, the systematic component of utility is assumed to be dependent on the attribute levels in the choice options, but also often to individual's socio-economic characteristics and the features of the choice task itself (Kragt and Llewellyn2014).

Further assuming that a particular alternative will be chosen only if it provides greater utility than any other alternative, and that the probability of choosing an alternative will increase if the quality or quantity of a 'good' attribute rises, a range of econometric techniques can be used to estimate the preference parameters for the random utility model from CE survey data. Alternative techniques reflect different assumptions about the nature of the error term (unobserved utility component) and the heterogeneity of preferences across respondents. Most common in the CE literature are the multinomial logit model and the latent class model. In terms of informing the TEV approach to economic valuation, preference parameters can then be used to estimate the willingness to pay for incremental (or marginal) changes in the level of individual attributes by dividing the estimated parameter for the attribute of interest by the estimated parameter on the cost attribute.

The CE method originates from the marketing literature where it has been used to analyse consumers' choices of products to inform trade-offs between multiple characteristics of goods (Louviere et al. 2000). It has also been used extensively in transport (Caussade *et al.* 2005), health (Ryan *et al.* 2007) and environmental management (Hanley *et al.* 2001). Given its ability to value individual ecosystem attributes and dimensions of management, it is now the most popular stated preference method for measuring non-use values in the marine environment.

Jobstvogt et al. (2014) use CE to estimate the value of protecting deep-sea biodiversity in Scotland through the creation of additional deep-sea marine protected areas. Three attributes were considered: number of species, new medicine products (potential discoveries from deep-sea organisms) and a cost (per household, per year). The results indicated high willingness to pay for deep-sea protection. Börger et al. (2014) also use a discrete CE to assess to what extent the general public perceive and value conservation benefits arising from an offshore MPA on the UK part of Dogger Bank in the North Sea. Based on real-world management options for fisheries, wind farms and marine protection in the area, the survey data revealed significant values for species diversity, the protection of certain charismatic species and a restriction in the spread of invasive species across the site. In Australia, Burton et al. (2014) use a choice experiment to assess the effects of various information treatments on the public's willingness to pay to afford greater levels of protection to five key features of the South-east Australian Marine Parks Network, namely bioregions, seafloor types, key ecological features, areas less than 1500 metres deep, and areas important to white shark populations. While the manner in which information was delivered to respondents in the survey did not affect preferences, analysis revealed three distinct types of respondents, with individuals' opinions about the importance of protecting sharks and whales being a significant determinant of class membership and hence preference structure.

CE's have also been used to estimate values or preferences for aspects of marine management, thereby addressing a key challenge in choice experiments -that being to frame the experiments in ways that reflect the policy issues to be addressed. Glenn et al. (2010) focused on fishing activity allowed in MPAs, the MPA strategy to protect corals and the cost of management to inform MPA design to protect deep sea coral reefs in the Republic of Ireland. Chhun et al. (2015) use the results of a choice survey in New Zealand for four socio-ecological attributes of the marine environment to estimate the publics' willingness to pay, in terms of higher taxes, for several alternative marine management scenarios. They identify a willingness to pay on the part of the public for management scenarios that impose restrictions on fishing to achieve higher levels of biodiversity and Maori cultural practices, and argue for the inclusion of such information in formal cost benefit analysis of policies such as marine parks.

Rogers (2013) used a discrete choice experiment to estimate how the community values the ecology of the Ningaloo Marine Park, with a view to understanding the drivers of social welfare in relation to marine conservation. A novel aspect of this research is that it not only considers the values people hold for conservation outcomes, but also their preferences for how those outcomes are achieved. The results indicate that management process does have an impact on individuals' preferences for conservation. Rogers (2013) also uses the CE method to identify differences between the preferences for conservation outcomes in two Western Australian marine parks between expert marine scientists and the general public. One case of an apparent divergence of preferences is partly explained by lack of public awareness, suggesting the importance of public education campaigns and care in basing policy on individual public preferences.

In a novel application outside the marine environment, Kragt and Llewellyn (2014) used a CE to estimate the values that Australian farm advisers attach to specific attributes of decision support tools (cost, specificity, input time and accuracy) relating to weed and herbicide resistance management, finding distinct 'market segments' among users. While CE's have largely evolved as a means of generating monetary economic values, such surveys can also be used more generally to inform decision-making in a multi-criteria decision context (see section 2.2.2.3), by focussing on trade-offs between non-monetary attributes. Jennings et al. (2015) examines the public's preference for attributes of shorebird offsets in Queensland, finding willingness to accept offsets that created benefits away from the impacted site so long as a greater number of birds were protected. Evidence was also found of the public's willingness to trade-off protection of the impacted species for protection of a more endangered species (i.e. trading-up). This result was consistent with the findings of a comparable study of seagrass offsets in Western Australia (Rogers et al. 2014).

The *contingent valuation* (CV) method establishes a hypothetical market for an environmental good or service and uses a survey to elicit peoples' willingness to pay (WTP) for some positive change in the supply or quality of the good or service (or willingness to accept in the case of a negative change). In the CV method, the value of an environmental good is elicited through a direct question(s) about their preferences. In short, this technique offers the respondent a choice between maintaining the status quo of a good or paying for a change in provision of the good. There are a number of possible formats for the direct willingness to pay question, ranging from an open-ended question asking simply for respondents to identify their maximum willingness to pay for the good; to dichotomous choice formats, where respondents are asked whether they would be willing to pay a specific dollar amount for an environmental change, with the amount varied across respondents. Greater sophistication in the question format requires more sophisticated econometric techniques to reveal estimates of economic value. In the simple case of the open-ended format, mean and median estimates of willingness to pay can be calculated from responses directly; whereas for dichotomous choice format data more complicated econometric techniques are required.

Dichotomous choice response data are generally analysed based on the random utility model, on the assumption that a ‘yes’ response from a respondent means that the utility of the changed quantity or quality of the environmental good, net of the payment amount, exceeds the utility of the status quo. Sample wide estimates of willingness to pay for the environmental change can then be calculated from the estimated parameters of the model.

In a bibliography of CV studies worldwide Carson (2011) identify over 7500 studies and papers from over 130 countries. In addition to traditional applications valuing air and water quality improvements, they note numerous applications in transportation, and growing use in health-related valuations and the valuation of cultural amenities. Examples of CV in the marine context in Australia include Yamazaki et al. (2013) who use a dichotomous choice CV survey to estimate the value of a day’s fishing for two important recreational fisheries in Tasmania. Gillespie and Bennett (2011) elicited the WTP for increasing the coverage of Commonwealth MPAs in the South-West of Australia by 10, 20 or 30% using both dichotomous choice and open-ended question formats. The latter were found to be systematically smaller than the former, but both methods found no significant difference between willingness to pay for 10% and 30% protection. One explanation for this is that respondents may consider 10% to comprise a large protected area and additional areas may contribute little to utility.

Benefit transfer

Benefit (or value) transfer (BT) is the use of information from existing primary study sites to estimate preferences and/or welfare changes at other unstudied or policy sites. The overriding advantage of benefit transfer is that it allows estimates of ecosystem service value to be included in decision analysis in cases where resource limitations or other constraints preclude primary research.

Benefit transfer generally takes one of four forms. In the first form, a single unit value from one or more studies (such as average willingness to pay per whale sited or per square kilometre of seagrass habitat) is applied to the analysis of other sites. The second form of BT involves making simple adjustments to the unit value to reflect differences between the study and policy sites, such as for differences in household income levels. A third type of BT uses value or demand functions estimated in primary studies (such as travel cost or contingent valuation studies) in conjunction with information on policy site parameter values to transfer benefits. Meta-analytic function transfer is a fourth form of BT where a value function is estimated from multiple primary studies, thereby allowing for greater variation in both site and study characteristics to be accounted for. A particular form of meta-analysis, known as structural benefit transfer, maintains a link to economic theory by defining a utility function for a representative agent and uses the outcomes of valuation studies to calibrate preference coefficients (Weber 2015).

While BT offers a practical solution to the need to provide information about potentially impacted values at unstudied policy sites, transferred values are subject to a number of errors. Specifically, errors in BT may arise from (i) errors in estimating original values at studied sites, (ii) transfer of values to policy sites that are different from study sites without adequately adjusting estimates to reflect these differences (generalisation errors),and (iii) publication selection bias. Some evidence suggests that the potential for transfer error is lower with value function and meta-analytic function transfer methods, however in cases where a high-quality study site exists for a policy site, simple unit value transfer may be adequate.

In addition to understanding and considering the nature and extent of potential transfer error in BT, practitioners must also pay careful attention to a range of issues related to the aggregation of transferred values across individuals and/or areas to get an estimate of total value for a particular ecosystem service, and across a number of different ecosystem services to get an estimate of the value of a particular ecosystem. Issues related to the spatial scale of transferred issues are also important, as ecosystems vary in both spatial scale, and in the geographical scale over which they provide services. The beneficiaries of these services also vary in terms of their location relative to such services. Further challenges exist in scaling-up transferred values in BT to consider the value of entire regional ecosystems, where marginal values may not be constant and non-linearities in ecological dynamics, particularly for critical services, may exist. Johnston *et al.* (2015) provide references on contemporary benefit transfer methods, debates, applications, challenges and frontiers, for both practitioners and users. Akter and Grafton (2009) describe a decision heuristic to help determine whether BT should be used, which method to use and how to account for transfer error in conservation decision-making.

Despite its many limitations, BT remains the most widely practiced approach to providing non-market valuation among policy makers and a number of publicly accessible databases exist to support the method's use. The Ecosystem Service Valuation Database, for example, is a relational database comprising 1,350 value estimates, which makes it possible to easily extract valuation data by valuation method, biome and ecosystem service (deGroot et al. 2012). For example, Costanza *et al.* (1997) estimated the monetary value of the contribution of the world's ecosystems to human well-being using a unit value benefit transfer methodology. In Costanza *et al.* (2014) their estimates were updated, generating an estimate of the value of the aggregate annual flow of ecosystem services provided by open ocean and coastal ecosystems globally of about $49.7 trillion (in 2007 dollars) in 2011. van de Belt and Cole (2014) used the same rapid ecosystem services assessment method to estimate the value of 55 ecosystem services across 8 biomes (including open sea/ocean and the continental shelf) for seven case study areas, including the EEZ, a mammal sanctuary and five marine parks in New Zealand. The commercial and non-commercial values of the natural resources of the Victorian coast were estimated using BT in WorleyParsons (2013), including those for beach recreation, seagrass, mangroves and other marine habitat types.

In more localised studies, Weber (2015) compares the total economic value of doubling the size of the Willamette Spring Chinook salmon run using seven different BT models and finds a range in values from $46.41 to $4,370.83 per household. In Australia, Raybould and Lazarow (2009) draw on benchmark primary US studies to estimate the total beach value associated with tourism. Allens Consulting Group (2009) use BT to provide estimates on displaced recreational fishing and the environmental non-market value of the stock of marine sanctuaries in the South-West marine region.

Deliberative and other non-monetary valuation

The wide array of methods to elicit economic values of the environment (as described above) have well documented practical limitations and are also often challenged on the basis of their strong foundation on the neoclassical economic conception of value. Specifically, that individual's behaviour and hence estimated values reflect rational, pre-formed, utilitarian preferences; the value of something to society is the aggregate of individual values; and that preferences are measurable (albeit indirectly through willingness to pay). In environmental management decisions it is increasingly recognised that many different dimensions of value are important. These include; transcendental values such as fairness, respect, and social status; contextual values which reflect views about specific environmental choices; and value indicators which are a measure of importance (e.g. willingness to pay and rankings). It is also acknowledged that individuals are but one source, or provider, of value, which should include *ad-hoc* groups (e.g. recreational fishers), communities (coastal fishing towns) and society as a whole. There is an increasing recognition of the role of deliberative and non-monetary approaches to both address the limitations of traditional economic valuation and to capture the broader suite of individual and shared values (Kenter 2015).

Deliberative Monetary Valuation (DMV), for example, refers to a range of approaches that integrate participation, reflection, discussion and social learning processes into monetary valuation of the environment, combining economic and political processes as part of the value-articulation institution (Lo and Spash 2013). In essence, small groups explore their values and preferences for different policy options. The end point can be deliberated preferences, whereby group members express individual values (e.g. choice experiments)via a process of deliberation aimed at assisting individuals in the formation of considered and coherent preferences. The end point can alternatively be deliberative democratic monetary valuation (DDMV), in which the group establishes a shared monetary value, in the form perhaps of an arbitrated social willingness to pay. Monetary values used in INFFER are based on a deliberative expert-based process where environmental values are elicited from environmental managers using a table of well-known environmental assets as examples and a scoring system that converts these to dollar values (Pannell *et al*. 2012). Lo and Spash (2013) caution that the purpose of deliberative approaches should not be seen as attempts to make incompatible value positions compatible, or to achieve accuracy of stated values, but rather to establish the legitimacy of value articulation in cases of contested economic, social and environmental values.

In addition to analytical DMV methods, Kenter (2014) list a wide range of other deliberative techniques such as citizens' or community juries and in-depth discussions, which generally result in qualitative outcomes such as lists, recommendations and verdicts. For instance, Ogier and McLeod (2013) use deliberative methods to identify marine values in the lower Huon Estuary and D'Entrecasteaux Channel, identifying seventeen key ecological, economic and social values ('that which you hold to be important about the marine environment') held by different communities with an interest (or stake), that are likely to be impacted by aquaculture. A variety of interpretive techniques (such as storytelling and participatory mapping sessions) and social-psychological methods (including subjective well-being indexes and values compass methods) can also be used in understanding and articulating transcendental and contextual values. Using five case studies of North American marine commercial fisheries, Courtland *et al.* (2010) illustrate an approach to representing subjective and objective measures of well-being.

While CE's remain the main survey-based approach to estimation of economic values, other survey-based approaches can be used to explore preferences. Increasingly popular among these is Best Worst Scaling (BWS), in which survey respondents are shown a subset of items from a master list and are asked to indicate the best and worst items (or most and least important, or most and least appealing, etc.). The task is repeated a number of times, varying the particular subset of items in a systematic way. Analysis is typically conducted, as with discrete CEs more generally, assuming that respondents make choices according to a random utility model, whereby an estimate of how much a respondent prefers item A over item B is provided by how often item A is chosen over item B in repeated choices.

Rudd (2014) uses this method to inform ocean research priorities, developing a ranking of 67 research questions based on a BW survey in which a global sample of research-active scientists were presented with 36 choice tasks, each of which asked them to indicate the most and least important among subsets of four research questions. Davis *et al.* (2015) used a seven attribute, seven choice set task (each comprising four options) to determine why fishers may choose not to participate in enforcement through monitoring of their exclusive territorial user rights in Central Chile. In a novel terrestrial example, Loureiro *et al.* (2012) use a two-stage BWS and latent class modelling to determine the preferences of forest owners in Spain for the social, economic and environmental consequences of alternative policies to prevent wildfire. They conclude that BWS may be a very suitable method of elicitation of preferences in the context of decision making under multiple conflicting criteria.

Multiple and hybrid valuation approaches

Economic valuation studies often use different valuation approaches to elicit values of different components of TEV (e.g. use and non-use values) and of the various benefits of ecosystem services. However, accepting value as a pluralistic concept in environmental management also implies the need to choose a valuation method that integrates a number of different methods. As an example, Kenter et al. (2013) contrast conventional CV estimates of the cultural ecosystem services of potential MPAs to British divers and sea anglers from an online survey, with shared values elicited through a consensus process via a series of deliberative workshops. They also demonstrated the advantage of combining information about monetary and non-monetary values (including story-telling, deliberative multi-criteria analysis and subjective well-being analysis) in providing decision-makers with information about both contextual and transcendental values. The authors suggest that the most appropriate method will depend on 5 keys things: (i) whether the proposed policy or management that is to be evaluated is likely to lead to significant conflict, or contestation, (ii) the complexity of the system under consideration, (iii) the services under consideration and values one might expect to find, (iv) practical limitations, and (v) the stage of the policy cycle.

There is also increasing focus on the integration of various decision-tools and valuation approaches into complex, hybrid conceptual frameworks to guide environmental management. Kompas and Liu (2013), for instance, develop a process for designing a valuation/decision framework that incorporates elements of both BCA and MCA. In a case study of biosecurity management options in Australia, they apply the process to demonstrate an approach that combines choice modelling and participatory MCA.

2.2.2 Tools and techniques

2.2.2.1 Cost-effectiveness analysis

Cost effectiveness analysis (CEA) is a means to compare alternative treatments, projects or activities when some non-monetary value for the desired outcome can be estimated, along with the costs of producing those outcomes. By assessing and ranking the relative performance of different treatments (or combinations of treatments) on the basis of the ratio of their costs and effectiveness, or outcomes, CEA can be used to identify which alternative among a restricted set of alternatives will achieve a pre-determined desired outcome or objective at the lowest cost. For example, if 10% of a given habitat-type is to be reserved, but there is little knowledge of the relative benefit of reserving one part versus another, CEA can be used to answer the question of which 10% can be reserved at least cost (Holland 2005).

In practice, CEA may involve simple spreadsheet analysis of discrete treatment/project alternatives; or complex mathematical programming formulated as either a cost minimisation problem (achieving exogenously determined target at least cost) or a benefit maximisation problem (achieving maximum benefit from a given budgetary outlay) (Balana et al. 2011). Regardless, CEA implies ranking by average cost per unit of effectiveness, or outcome.

The practice of CEA reflects multiple influences, having emerged independently from economic theory, the analysis of public investment in water resource development and as part of the application of operations research to problems of military development in WWII (Quade, 1971). CEA is now most widely used in the health care sector when comparing alternative treatments for a given condition (e.g. Price *et al.* 2005). The development of generalised health outcome measures has also resulted in its use for prioritisation of health care at a national level (e.g. Hutubessy *et al.* 2003; Devlin and Parkin 2004).

Applications of CEA in the marine environment are common and cover a wide range of management problems. For instance, Mumby et al (1999) use CEA to compare remote sensing and field survey methods for coastal habitat mapping, using a measure of overall map accuracy to define effectiveness. Pascoe et al (2011b) compare rodent control and fishing area closures as alternatives to offset seabird bycatch, based on average cost per additional seabird relative to a base case. Other applications include comparing risk control options aimed at mitigating the environmental risk of accidental oil spills (Vanem et al. 2008); and various structural and technological changes within sectors and activities aimed at reducing nutrient pollution of the marine environment (Paaby et al 1996). Tisdell *et al.* (2011) compare management strategies to control the range-extending long-spined sea urchin *Centrostephanus* in Tasmania, measuring effectiveness in terms of a pre-determined target urchin density. CEA of alternative mitigation measures is a formal requirement of the European Water Framework Directive in formulating programmes of measures. In a review of the Directive's progress, Balana *et al* (2011) cite several studies of the method's use in comparing national and international measures to reduce nutrient enrichment of the Baltic Sea.

While many applications of CEA focus on a single outcome, Westmacott and Rijsberman (2000) produce a range of cost-effectiveness measures of coral reef (i.e., rugosity, coral cover and area of reef lost or gained) for a range of both individual environmental protection measures and composite protection strategies in the Maldives. The authors note however that while multiple measures provide potentially valuable information for reef managers, in the absence of a decision rule for trading off potentially conflicting outcomes, managers may resort to a focus on costs only.

Joseph *et al.* (2009) build on traditional CEA in the context of allocating conservation funds to threatened species conservation, developing a nine step Project Prioritization Protocol (PPP) that simultaneously considered costs, benefits (including species value and uniqueness) and likelihood of success in a single composite measure of cost-effectiveness. In a New Zealand application of the PPP, they demonstrate the benefit of adopting a cost effectiveness rule in terms of total conservation benefit compared to more limited decision criteria based solely on measures such as threat status.

Other recent applications of CEA in conservation management have also moved the practice from the domain of the 'desktop analyst' whose role is to provide information to a decision-maker, to one that is embedded within a structured decision making (SDM) process. SDM is designed to engage decision-makers, stakeholders and scientists in the decision-making process, thereby incorporating scientific facts and values throughout. While the use of cost effectiveness in this way (i.e., using a composite measure of cost-effectiveness and implementing the analysis within an SDM process) has been more common to date in the terrestrial than marine domains (e.g. Cullen *et al.*2005), examples of the latter are increasing. Addison and Walshe (2015) use a similar method to inform the prioritisation of management interventions for investment of Reef Trust funds to protect values of the Great Barrier Reef World Heritage Area, drawing on experts to provide estimates of consequence including an indication of their uncertainty in their judgements.

2.2.2.2 Benefit-cost analysis

In cases where all expected benefits can be expressed in the same units as costs, i.e. they can be assigned a monetary value, benefit-cost analysis (BCA) is applicable. Expressing the relationship between benefits and costs in terms of a benefit-cost ratio lets us determine whether a particular project is worthwhile (relative to a base case); or, by ranking several alternative projects in terms of their benefit-cost ratios, to identify the best option. Selection of the alternative with the highest benefit-cost ratio has a strong basis in public policy and welfare economics. BCA is a systematic framework for economic appraisal of proposed projects from the point of view of the public interest and is equally applicable to public and private sector projects. BCA applies rational choice theory and its methods are deeply rooted in the neoclassical welfare economics paradigm. Specifically, BCA is a formal test of whether a project is efficient, in the sense that the amount by which those who gain from the project is large enough to (hypothetically) compensate those that incur losses, and for the gainers to still be better off. Many of the criticisms of BCA can be traced to the strong neoclassical economics assumptions and utilitarian values embedded in its methods (Anderson *et al.*2014).

The need to enumerate all impacts of a project that affect economic welfare in terms of a common dollar metric can be challenging. In essence, BCA compares the value of the outputs produced by a project to the value of the output that could have been produced by the resources used in the project in their best alternative use (i.e., their opportunity cost). While economic theory suggests that market prices for both project inputs and outputs reflect such values, and could therefore be used to measure benefits and costs, two considerations are often limiting:

1. The actual (revealed) prices in many input and output markets are often not an accurate reflection of true social values due to a range of factors including the presence of imperfect competition and distortions associated with measures such as taxes and tariffs. In such cases, market prices must be ‘corrected’, and shadow-prices used to ensure the proper valuation of costs and benefits. While accepted methods for shadow-pricing inputs and outputs exist (Campbell and Brown 2016), they generally require technical skills and may not be easy to explain to those with a stake in the outcome of the analysis.

2. Many inputs and outputs of projects, particularly in the case of decisions involving the use of natural resources and environmental assets, are not traded in markets. In cases where no markets exist, BCA practitioners must rely on values generated by the preference elicitation methods described above, but the time and resources required to apply these methods are substantial. In any case, stakeholders are unlikely to feel comfortable with monetisation of all objectives, especially those dealing with social and environmental outcomes. The notion that individual preferences are able to provide a complete representation of the social values associated with protecting environmental assets is also increasingly questioned (Anderson *et al.*2014), further challenging the individualistic, utilitarianism that underpins BCA.

A key feature of BCA (and CEA) is the manner in which it accounts for the fact that decisions involving the allocation of resources invariably involve consequences (costs and benefits) that occur over time, making it difficult for managers to compare outcomes. In BCA these are adjusted using discounting, a process whereby costs and benefits occurring in different time periods are converted to their present value, with future consequences reduced to reflect a preference for more immediate benefits. The selection of the appropriate rate at which to discount future consequences (relative to those occurring now) can be controversial and is the subject of much debate, particularly as it relates to environmental management. High discount rates tend to favour projects or management options that yield benefits relatively quickly but for which costs can be spread over time; whereas those involving immediate investments and with benefits in the distant future are disadvantaged. Economic theory provides some guidance to choosing the discount rate, suggesting it should reflect both the opportunity of public sector funds used and the rate at which society is willing to trade-off current for future consumption. Based on the assumption that it reflects both these things, the real rate of return on long term government bonds (currently around 3% in Australia) is often recommended in public sector project evaluation. It is worth noting that even at the relatively low rate of 3%, a project that involved spending $1000 today would need to produce an outcome worth more than $19k in 100 years to be preferred over the base case (or do nothing) alternative using the benefit-cost criteria.

While primarily efficiency-focussed, BCA can also account for the distributional consequences of alternatives, for example by weighting losses and gains experienced by different income groups, sectors, or regions differently. In practice however, the application of distributional weights is uncommon. Campbell and Brown (2016) recommend that in addition to measuring aggregate net benefits, that the BCA include a referent (or stakeholder) group analysis to provide the decision-maker with information about the distribution of net gains and net losses.

Compared to CEA, BCA has the advantage of being able to judge the economic efficiency of a proposed action; whereas the focus of CEA is on which of a group of alternatives is best at achieving an outcome, not whether the outcome itself is worth achieving at all. In addition, BCA can be used to compare projects across vastly different outcomes or even across different areas of public policy; whereas CEA is restricted to comparing actions that are aimed at achieving the same broad objective. Nevertheless, since a reduced number of impacts need to be valued in CEA it is generally less onerous than BCA; and its avoidance of the need to monetize all costs and benefits, and to accept the assumption that the whole of value is equal to the sum of individual preferences may be appealing.

That said, there is a strong tradition of BCA in public policy analysis both internationally and in Australia, with BCA being generally promoted as the principal method informing major public policy, regulatory and project-level decisions across all levels of government. While the data and technical demands of a full BCA can be demanding, particularly for projects involving a large number of non-market impacts, a preliminary process of project scoping can assist managers form a judgement about which decisions warrant serious assessment. In essence this informal benefit-cost analysis can contribute to ensuring society gains the benefit associated with undertaking an economically good project or avoids the economic loss associated with undertaking an economically bad project.

2.2.2.3 Multi-criteria analysis

Maguire (2004) cites two interacting flaws commonly encountered in decision-making under uncertainty: (a) incoherent treatment of the essential connections between social values and the scientific knowledge necessary to predict the likely impacts of management actions, and (b) relying on expert judgment about risk framed in qualitative and value-laden terms, inadvertently mixing the expert’s judgment about what is likely to happen with personal or political preferences. The family of techniques under the banner of MCDA seek to avoid these flaws through explicit separation of the task of causal judgment from the task of articulating value judgments or trade-offs (Ananda and Herath 2009). Here we outline several of the many members of the family, including:

- Multi-attribute Value Theory
- Analytic Hierarchy Process
- Outranking
- Multi-attribute Utility Theory

MULTI-ATTRIBUTE VALUE THEORY (MAVT)

The technical description of MAVT that follows is adapted from Bedford and Cooke (2001) and Keeney (2007).

The task of MAVT is to find a simple expression for the decision-maker’s value function v over two or more relevant attributes (i.e. objectives and associated criteria). The additive value model is commonly used, in the form

$$\mathrm{v}(x_1,\ldots,x_n) = \sum_{i=1}^{n} w_i v_i\left(x_i\right)$$

where the w_i are the weights and the v_i are marginal value functions.

A marginal value function is a value function for any single attribute in isolation. A formal way of eliciting a marginal value function is as follows. Suppose that we want to determine a value function for x_1. Write the vector of attributes exclusive of x_1 as $\underline{y} = (x_2,\ldots,x_n)$. We can pick two values for the attribute x_1, say $l<h$, and arbitrarily assign $v_1(l) = 0$ and $v_1(h) = 1$ (assuming that lower values of the attribute are worse than higher values). We now want to interpolate and find a number $m_{0.5}$ between l and h so that $v_1(m_{0.5}) = 0.5$ (Figure 15). To do this we pick a value for the other attributes, $\underline{y}$, and seek a ‘worse’ value for the other attributes $\underline{y}'$ so that for some $m_{0.5}$ between l and h,

$$(l, \underline{y}) \sim (m_{0.5}, \underline{y}'), \text{ and}$$
$$(m_{0.5}, \underline{y}) \sim (h, \underline{y}').$$

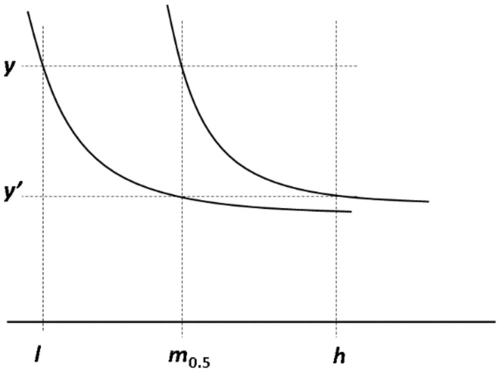


Figure 15. Formal elicitation of a single attribute value function.

Writing $v_{\underline{y}}$ for the weighted sum of the value functions in $\underline{y}$, we then have

$$v_1(l) + v_{\underline{y}}(\underline{y}) = v_1(m_{0.5}) + v_{\underline{y}}(\underline{y}'),$$
$$v_1(m_{0.5}) + v_{\underline{y}}(\underline{y}) = v_1(h) + v_{\underline{y}}(\underline{y}'),$$

which together gives $v_1(m_{0.5}) = 0.5$. In this (laborious and cognitively demanding) way we can interpolate the value function for as many points as desired. The same procedure is required for each attribute.

A common simplification is to assume linearity between $v_1(l) = 0$ and $v_1(h) = 1$. It avoids the tedious demands of formal elicitation and is reasonable over the local range of consequences associated with most problems (Durbach and Stewart 2009). There are two good reasons for this assumption. First, some objectives only make ethical sense as linear value functions. For example, imagine a catastrophic loss of 200 lives. There is no ethical basis for valuing the loss of the 200th life any differently to the loss of the first life. Second, linearity is reasonable for a single decision when many other decisions and influences affect the same objective (Keeney and von Winterfeldt 2007).

Having obtained marginal value functions, we need to weight them. This can be done formally by the method of indifferences, akin to the underpinnings of stated preference techniques used in evaluation of non-market impacts in benefit-cost analysis (Bennett and Blamey 2001). Suppose that x_1 and x_2 are the first two attributes, and that $\underline{b}$ is the vector of remaining attributes. Let x_1^* and x_2^* and $\underline{b}^*$ be the attribute values for which the marginal value functions are zero. Then if we can find values $x_1 \neq x_1^*$ and $x_2 \neq x_2^*$ such that

$$(x_1,\ x_2^*,\ \underline{b}^*) \sim (x_1^*,\ x_2,\ \underline{b}^*)$$

then $w_1 v_1(x_1) = w_2 v_2(x_2)$. Proceeding this way, we can get $n - 1$ linear equations relating weights (without loss of generality we can assume the weights sum to 1), and solve for the w_i. Again, the method is laborious and cognitively demanding.

There are many shortcut methods for eliciting weights, each with advantages and disadvantages (Doyle *et al.* 1997; Bottomley *et al.* 2000; Hayashi 2000; Hajkowicz et al. 2000; Bottomley and Doyle 2001; Roberts and Goodwin 2002; Wang *et al.* 2009). Comparative studies of these methods suggest in some cases that the weights may vary considerably (Pöyhönen and Hämäläinen 2001), although others have found higher correlations between the methods (Van Ittersum *et al.* 2007).

Three commonly used approaches to determine preferences include simple ranking approaches (Roberts and Goodwin 2002), scoring based approaches (Bottomley and Doyle 2001) and the Analytic Hierarchy Process (AHP) (Saaty 1980) based on a series of pair-wise comparisons (see description of AHP below). Each method relies on a selected group of individuals (e.g. key stakeholders) to indicate a preference for each objective within a set of objectives. They differ in how these preferences are captured and analysed, both between and within the different approaches.

Ranking based approaches require individuals to simply rank sub-components from most important (with a rank of 1) to least important (with a rank of *n* where *n* is the number of sub-components compared). Examples of ranking based systems in coastal and natural resource management applications include Huylenbroeck and Martens (1992), Heilman *et al.* (1997) and Sheppard and Meitner (2005).

Scoring based methods, or direct rating methods, involve allocating a score, for example 100, to the most preferred (first ranked) sub-component, then allocating a lower score somewhere between 1 and 100 for subsequent sub-components based on their relative importance. Direct rating methods have been applied in a number of coastal and resource management studies (Yang *et al.* 2011; Koschke *et al.* 2012; Liu *et al.* 2013).

As with ranking approaches, there have been several different approaches proposed for weight derivation from scoring based systems. These include direct rating approaches such as the max100 approach, where the highest ranked sub-component is allocated 100 points and subsequent (lower) sub-components allocated less than 100 points; and the min10 approach where the least preferred sub-component is allocated 10 points and the higher ranked sub-components allocated higher points relative to these (Bottomley and Doyle 2001). Alternative approaches include direct point allocation where the set of all sub-components are allocated 100 points, and individuals share these 100 points out across all sub-components (so that they sum to 100) (Bottomley and Doyle 2001). In all cases, the final weight is determined by:

$$w_{i,j} = S_{i,j} / \sum_{i} S_{i,j} \tag{2}$$

where $S_{i,j}$ is the initial score given to each sub-component *i* (i.e. between 1 and 100) and $w_{i,j}$ is the final weight used in the analysis.

Several studies have suggested that the direct rating approach involving setting the higher ranked sub-component a score of 100 is the most reliable in test-retest studies (Doyle et al. 1997; Bottomley et al. 2000; Bottomley and Doyle 2001). von Winterfeldt and Edwards (1986) and Fischer (1995) consider the swing weight method (a scoring-based approach) to be one of the more effective, in terms of its efficiency, practicality and its insulation against abuse.

Whatever method is used in their elicitation, the interpretation of the weights is critical. Methods that do not explicitly deal with indifferences are prey to incoherence and inconsistency. Users are inclined to specify weights that reflect the relative importance of the attributes, irrespective of the units or the range of consequences relevant to the decision context. But the weights have units because the underlying attribute scales have units. A change of $-w_i^{-1}$ units on scale *i* is always compensated by a change of $+w_j^{-1}$units on scale *j*. Changing the units or range of an attribute *must* lead to a change in the weights.

Keeney (2002) asked a group of television and newspaper reporters involved in coverage of hazardous waste sites to rank the importance of (a) economic costs of site remediation, (b) public health outcomes, and (c) environmental harm. No information on the magnitude of impacts was provided. Of the 80 respondents, 79 ranked public health highest. Keeney then asked the same group to rank the importance of a $3 billion expenditure, avoiding a mild two-day illness to 30 people, and destroying 10 square miles of mature forest. For most respondents, the rank order of the three considerations was the exact opposite of the context-free framing. Any weighting technique that fails to promote normative interpretation of weights through explicit consideration of the range of consequences is inadequate (Steele et al. 2009).

For the *additive* value model commonly used in MAVT to be valid the attributes need to be *mutually preferentially independent*. That is, the value ascribed to any given amount of attribute *i* cannot be conditioned by the level available of attribute *j*.
In practice, the assumption of preferential independence is reasonable if the set of objectives is consistent with the following properties (Keeney 2007):

- Complete – all of the important consequences of alternatives in a decision context can be adequately described in terms of the set of fundamental objectives.
- Non-redundant – the fundamental objectives should not include overlapping concerns.
- Concise – the number of objectives should be minimal.
- Specific – each objective should be specific enough so that consequences of concern are clear, and attributes can readily be selected or defined.
- Understandable – any interested individual knows what is meant by the objectives.

Where objectives satisfy these properties there is a strong case for use of simple weighted summation. While the analyst needs to be careful to ensure preferential independence, the mechanics of MAVT are straight-forward. Arithmetic operations are simple and easy to implement in a spreadsheet.

Strictly speaking, MAVT is applicable where there is no uncertainty in the estimation of consequences or where decision-makers and stakeholders can be assumed to be risk-neutral. These assumptions are unrealistic in many contexts.

ANALYTIC HIERARCHY PROCESS (AHP)

AHP is commonly encountered in MCDA applications in the natural resource management literature (Mendoza and Martins 2006). It is essentially a variant of MAVT designed to minimise the elicitation burden on experts and decision-makers. Most applications employ the same additive value model described above for MAVT. Using a nine-point preference scale and matrix computations to translate ordinal judgments into cardinal judgments, (a) marginal value functions and (b) weights, are derived through pairwise comparisons of alternatives and objectives, respectively (Saaty 1980). A variety of software packages are available, although for simple problems the calculations can be done in a spreadsheet.

AHP has been used in a number of marine and coastal applications to determine management sub-component importance and assist in decision making (Leung *et al.* 1998; Soma 2003; Mardle *et al.* 2004; Wattage and Mardle 2005; Nielsen and Mathiesen 2006; Himes 2007; Pascoe *et al.* 2009a; Pascoe *et al.* 2009c; Baby 2013; Pascoe *et al.* 2013), and is the most common approach used for preference elicitation in a wide range of applied natural resource case studies. The pair-wise comparison method makes the process of assigning weights much easier for participants because only two sub-components are being compared at any one time rather than all sub-components having to be compared with each other simultaneously. Preferences are expressed on a nine-point scale, with a 1 indicating equal preference, and a 9 indicating an extreme preference for one of the sub-components. Preferences are assumed symmetrical, such that if A against B has a preference of $a_{AB} = 9$, then $a_{BA} = 1/a_{AB} = 1/9$. For each set of comparisons, a matrix of scores can be developed, given by:

$$A = \begin{bmatrix} a_{11} & a_{12} & \dots & a_{1n} \\ a_{21} & a_{22} & \dots & a_{2n} \\ \dots & \dots & \dots & \dots \\ a_{n1} & a_{n2} & \dots & a_{nn} \end{bmatrix} \tag{3}$$

The scores are normalised by dividing through each element of the matrix by the sum of the column *j* (i.e. summed over *i*, such that $\overline{a}_{ij} = a_{ij} / \sum_i a_{ij}$), and the weight associated with each sub-component can be estimated as the average of the normalised scores across the row *i*. That is, $w_i = \sum_j \overline{a}_{ij} / n$, where *n* is the number of sub-components being compared.

AHP's strength in minimising elicitation burden is also its weakness. It's possible to obtain marginal value functions without any explicit estimation of consequences. For decision problems involving self-evident cause-and effect relationships this may be acceptable. For other contexts, the consequences of alternatives involve difficult probabilistic judgments that are likely to be logically challenging (Hastie and Dawes 2010), and poorly captured and poorly documented in pairwise comparisons.

AHP has also been criticised on theoretical grounds because it allows rank reversal upon introduction of a new alternative - a violation of decision theory's independence of irrelevant alternatives axiom[1] (von Neumann and Morgenstern 1944). The modified AHP (mAHP) is free of this problem. It uses standard MAVT techniques to obtain marginal value functions, and limits the use of pairwise comparisons to the derivation of weights. Moffett and Sarkar (2006) advocate use of mAHP because of the relative ease of obtaining weights. But weights obtained through pairwise comparisons via mAHP may result in poor capture of stakeholder preferences. In general respondents tend to assign weights according to the perceived importance of objectives, irrespective of the consequences associated with the specific alternatives being considered.

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[1] Here's an example of a violation of the independence of irrelevant alternatives axiom: Imagine a customer in a restaurant has a choice between two dishes - fish or steak. They order the fish. The waiter soon returns to inform the customer that a third option, pasta, has become available. When asked which of the three dishes they prefer, the customer changes their order to steak.

OUTRANKING

Outranking techniques stem from the French school of MCDA, which places less emphasis on normative understanding of how decisions *should* be made based on axioms of rationality (von Neumann and Morgenstern1944) and greater emphasis on behavioural models of decision-making (Roy 1973).

Outranking techniques typically involve sequential elimination of alternatives (Chankong and Haimes 2008). Weights are assigned to each objective according to their perceived importance, without consideration of the range of consequences associated with alternatives. For each pair of alternatives, a concordance index and a discordance index are constructed. The concordance index coarsely characterises the strength of the argument that one alternative is better than another based on the weighted sum of objectives for which it dominates the other. The discordance index reports the strength of the argument against eliminating the (weakly) dominated alternative. Decision-makers work through a consequence table iteratively, adjusting critical thresholds for concordance and discordance until a satisfactory choice is made.

There are numerous techniques and software packages that fall under the banner of outranking (e.g. ELECTRE, PROMETHEE, GAIA; see Figuera et al. 2005 for details). The techniques vary according to how expected consequences are characterised. If a consequence table is populated using qualitative ordinal descriptors of impact ELECTRE can informally assist stakeholders progress trade-offs and difficult decisions involving more than a handful of objectives and alternatives. While other outranking techniques can be used where consequence estimates are quantitative or semi-quantitative, there is little argument for doing so, because in these circumstances MAVT offers a much firmer normative basis for decision-making.

MULTI-ATTRIBUTE UTILITY THEORY (MAUT)

The formal description of MAUT developed by von Neumann and Morgenstern (1944) was (and remains) a high point in the theory of decision-making. It is also a wholly impractical approach to typical multi-objective, multi-stakeholder problems. Many of the developments and refinements of MCDA since the 1950s are essentially pragmatic short-cuts for MAUT.

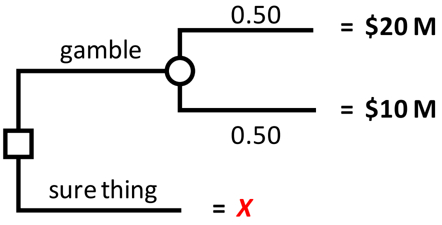
MAUT can be used when a consequence table is populated by statistical distributions describing probabilistic uncertainty in the performance of each alternative against each objective. These circumstances are rare indeed, especially in natural resource management. Aside from difficulties in obtaining detailed probabilistic causal judgments, there are distinctly onerous demands on decision-makers and stakeholders in the elicitation of trade-offs under MAUT.

MAUT describes the decision-maker's utility function *o*ver two or more relevant uncertain attributes. If preferential independence can be assumed the additive utility model is appropriate, in the form

$$u(x_1,\ldots,x_n) = \sum_{i=1}^{n} u_i k_i\left(x_i\right)$$

where the k_i are scaling constants and the u_i are marginal utility functions.

The elicitation of marginal utility functions is more complicated than the corresponding exercise in MAVT, because the risk preference of the decision-maker needs to be considered. A series of 'certainty-equivalent' questions are posed to enable a utility function to be fitted. An example is shown below using monetary payoffs. The task for the decision-maker (or stakeholder) is to assign a value for *x* such that they are indifferent to the choice between the gamble and the sure thing. If *x* = $15M in our example, the decision-maker is risk-neutral. Values of *x* less than $15M imply risk aversion. Values greater than $15M indicate a risk seeking decision-maker. Risk aversion is far more common than risk-seeking behaviour, but individuals vary in the extent to which they are risk-averse (von Winterfeldt and Edwards 1986).



To obtain scaling constants, an even more demanding series of questions are posed. In practice, only the most committed and indefatigable decision-makers are capable of formally addressing trade-offs using MAUT.

2.2.2.4 Combining methods – ‘qualitative’ management evaluation

Where management objectives can be identified, multicriteria approaches can be applied to rank or assess different management options. A method, adapted by Pascoe et al. (2009b), links management objective weightings (derived using AHP) to performance scores (on a scale of -3 for severely worse off to +3 for substantial improvement) derived through expert opinion to rank alternative management options. As well as the original study considering spatial management options, the approach has been applied to assess management options for the Queensland east coast trawl fishery against environmental, social and economic objectives. In this case, the outcomes under each management strategy were assessed from the perspective of the different stakeholders (using their identified preference structures), and the results compared (Dichmont et al. 2012; Dichmont et al. 2013). Similar approaches have also been applied to assess a range of terrestrial natural resource management options (Hajkowicz *et al.* 2000).

A comparable approach was also used to develop a social, cultural and economic report card for Gladstone harbour (Pascoe et al. 2014c). In this case, a BBN framework was used to combine the different indicators to the objectives. The conditional probability tables were derived based on expert opinion from a group of social scientists using multicriteria elicitation approaches previously discussed. Objective weightings were derived through preference elicitation approaches from both the general community and also the resource managers (for comparative reasons). Indicator scores were largely distributions of satisfaction against a range of criteria, collected from a survey of the Gladstone community. The outcomes of the approach were a score against each objective and also the broader objective domain.

2.2.2.5 Programming and optimisation

There are potentially thousands of alternative management strategies within marine park networks. Various mathematical programming techniques from the field of Operations Research are available to help identify better (or best) candidates.

Linear Programming (LP) and *Stochastic Dynamic Programming* (SDP) employ algorithms designed to optimise some objective function under specified constraints (Chankong and

Haimes 2008). For example, we might seek to minimise costs to fishing subject to constraints defining thresholds for minimum performance for conservation objectives. In LP, a static linear relationship (or near-linear) relationship between management action and expected consequences is required. This is clearly inappropriate in natural systems, where outcomes for objectives are characteristically dynamic and non-linear.

With detailed understanding of cause-and-effect, SDP can accommodate non-linear, dynamic outcomes. The capacity to capture greater realism in SDP is attractive, but computational overheads and the requirement for sophisticated causal understanding mean that most applications are substantially simplified.

The treatment of trade-offs in LP and SDP through specification of constraints is simplistic. An alternative either satisfies or does not satisfy the constraint. Constraints in optimisation problems can lead to demonstrably poor outcomes (Minin and Moilanen 2012) and have been criticised as an *ad hoc* treatment of preferences (Moffett and Sarkar 2006).

Goal programming (GP) avoids the naive binary logic of constraints. It involves specification of a performance aspiration for each objective. The underlying algorithm searches among the candidates for the alternative having the minimum multi-dimensional distance to the goal set (Chankong and Haimes 2008). Conceptually, the method could be used profitably by a single decision-maker. In a multi-stakeholder setting, GP is potentially open to abuse if stakeholders try to manipulate outcomes through articulation of insincere positions on what might be considered an appropriate goal for each objective.

The key factor limiting practical application of any of these techniques is the requirement for a mathematical function for each objective linking *all* possible alternatives and expected consequences. Simulation modelling may provide some insight into the form of these mathematical functions for some objectives. For other objectives our understanding of cause-and effect may be insufficiently mature to allow reasonable estimation of the form of the function and its parameters.

As our understanding improves, there may be opportunities to exploit the capacity of optimisation methods to search vast numbers of alternatives. One technique that steers a middle course between GP, LP and SDP is *simulated annealing*. Like problem formulation in LP and SDP, simulated annealing requires an objective function (to be minimised or maximised) and one or more constraints. But unlike LP and SDP, it interprets constraints as soft requirements. It uses a penalty function to downgrade the merit of an alternative proportional to the extent to which it fails to meet constraints, which is a more sophisticated treatment of trade-offs than that available in GP. It produces a set of better performing alternatives and can be used in an interactive exploratory manner with multiple stakeholders. Free software with the capacity to link with GIS is available (see http://www.uq.edu.au/marxan/).

Goal programming has been extensively used in fisheries analysis (Drynan and Sandiford 1985; Mardle et al. 2000a; Mardle et al. 2000b; Pascoe and Mardle 2001; Pascoe et al. 2011a) as well as other areas of natural resource management (Hayashi 2000). Similarly, SDP has also been applied in fisheries modelling (Kennedy and Pasternak 1991; Androkovich and Stollery 1994; Costello and Polasky 2008; Dowling et al. 2011).

2.2.2.6 Viability analysis

An alternative to optimising in a multi-objective modelling framework is viability analysis (Péreau et al. 2012). Viability analysis uses stochastic simulation approaches to determine the likelihood that a system, such as a fishery, will remain above some minimum acceptable level of each objective (e.g. minimum stock level, minimum profits etc.) under a given management strategy. Several examples of the approach have been developed for fisheries. For example, a model of the Northern Prawn fishery has been developed that focuses on economic and resource sustainability as well as an additional environmental objective to minimise bycatch of a protected species (Gourguet et al. 2016). Other examples include Béné *et al.* (2001); Eisenack *et al.* (2006); Martinet *et al.* (2007); Doyen *et al.* (2012); Péreau et al. (2012); Gourguet *et al.* (2013); and Sinclair (2014).

2.2.3 Adaptive management and learning

Adaptive management seeks to overcome the challenges of uncertainty through an emphasis on 'learning by doing'. In a very loose sense, historical use of trial and error implies that environmental managers have always practised adaptive management. The more formal interpretation of adaptive management comes from statistical control theory and its origins in manufacturing. Walters (1986) took the concepts from statistical control and applied them to fisheries management. Subsequent applications have embraced conservation management (e.g. Keith et al. 2011, Johnson et al. 2015). Although details vary, the generic elements are shown in Figure 16, which was developed for application in the management of Tasmania's terrestrial protected areas (Jones 2009).

Figure 16. The adaptive management cycle. Source: Jones (2009).

Figure 16 emphasises the role of evaluation and learning in informing 'adjustments' to management action - an intuitive idea, the appeal of which is magnified in systems characterised by limited understanding. Indeed, much of the research community in natural resource management asserts adaptive management as a universally applicable management system. For these researchers, the apparent failure to implement adaptive management in many settings is a source of considerable frustration (Williams and Brown 2014). But statistical control theory makes it clear that an adaptive approach should only be implemented where the value of the information gained through evaluation and learning is anticipated to be greater than the costs of acquiring that information (Walters 1986, Pratt et al. 1995). These circumstances may be less common than thought.

To illustrate, consider a choice between education and enforcement for improved outcomes for a species or ecosystem of conservation interest. Managers may be uneasy about spending scarce resources on our hypothetical species or ecosystem when there is a 30% chance its status is in fact non-threatened under the do-nothing scenario. Before committing to education or enforcement, Parks Australia may elect to commission research that clarifies the status of the species. This decision is an example of the allocation of resources for acquisition of information and improved understanding. The wisdom of delaying the (uncertain) management decision through investment in information acquisition can be considered through assessment of the value of that information.

First we estimate the pay-off of making a decision today under uncertainty. Let's say that managers value persistence in the distant future the same as the immediate future such that enforcement is perceived as a better option than education. The pay-off to enforcement versus do nothing under each uncertain state is shown in the decision table below.

	Conservation outcomes (EEY)		
	State = threatened, *p* = 0.70	**State = non-threatened, *p* = 0.30**	**Cost**
Do nothing	100 years	1000 years	$0
Enforcement	0.70 × 1 000 + 0.30 × 100 = 730 years	1000 years	$250k

This table makes the managers' unease plain. If we elect to go ahead today with implementation of enforcement there is a 30% chance we will spend $250k with no material conservation benefit. There is a 70% chance we will have 'purchased' an additional 630 years of persistence for our hypothetical species or ecosystem.

To calculate the expected pay-off of this gamble we need to articulate our preferences for monetary and conservation outcomes. Let's say that after some consideration, Parks Australia estimates that it is willing to pay $1k for each additional year of persistence for the species concerned. Now the table can be described in monetary terms.

The option with the highest expected pay-off, *if we were compelled to make the decision today*, is enforcement, at $561k. But we note that if the species is in fact non-threatened we will regret this decision.

	Conservation outcomes (EEY)		
	State = threatened *p* = 0.70	**State = non-threatened *p* = 0.30**	**Expected pay-off**
Do nothing	$100k	$1 000k	0.70 × 100 + 0.30 × 1000= $370k
Enforcement	$730 – $250k = $480k	$1 000– $250k = $750k	0.70 × 480 + 0.30 × 750= $561k

What is a fair price to avoid this regret? If we *knew* the species were threatened we would choose enforcement, with a value of $480k. If we *knew* the species were non-threatened we would choose do nothing, with a value of $1 000k. The expected value given perfect knowledge is 0.70 × $480k + 0.30 × $1000k = $636k. A fair price for avoiding the possibility of regret is $636k - $561k = $75k. That is, if we can fund a research program that will definitively tell us whether or not the species or ecosystem is threatened for $75k or less, we are better off delaying our management decision until the outcomes of that research. Note that this calculation of the expected value of perfect information is an upper bound on what we should be willing to pay for research, because no study can deliver perfect information.

Developed within the theory of information economics, value of information analysis can be viewed as a form of benefit-cost analysis for the special circumstance in which we are interested in assessing the merit of allocating resources for the acquisition of improved understanding. It has been applied to decision problems in diverse fields including medicine (Singh et al. 2008), epidemiology and health risk management (Shea et al. 2014), and mineral resource exploration (Eidsvik et al. 2008). Applications in environmental management have been rare in the past, but recent research suggests a growing interest in judicious application of adaptive management via an initial assessment of the value of information (Maxwell et al. 2015, Moore and Runge 2012, Williams et al. 2012).

3. IMPEDIMENTS TO ADOPTION

There has been a significant research effort into the development of decision support tools (DSTs) for marine resource allocation. For example, unpublished research by one of the co-authors of this report, Fiona Gibson and colleagues found that in the rezoning process for the Great Barrier Reef (GBR), undertaken by the Great Barrier Reef Marine Park Authority between 1999 and 2004 as part of the Representative Areas Program, the DST Marxan – a spatial prioritisation program for conservation planning – was used as the primary tool for engaging stakeholders and community members in the rezoning process. A second case study explored by Gibson and colleagues is the use of the Harvest Strategy Framework (HSF) in sustainable management of the Southern and Eastern Scalefish and Shark Fishery (SESSF). The researchers found strong uptake of this DST amongst the fishery managers, AFMA. However, the adoption of these tools in decision making is not a given.

There are several factors that contribute to the adoption of tools in decision making. In the GBR example, a number of factors were cited by the park managers as important in facilitating uptake of the Marxan DST: 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 outcome; the tool is able to deal with missing information; the policy process allows adequate time for the DST to be used; and the tool capabilities align with the objectives of the policy being implemented. In the case of the SESSF, the fisheries managers' responses aligned with the GBR marine park managers, except that they felt a champion for the decision support tool was not important in the adoption of the HSF.

Other studies in the marine resource literature provide further insight into DST adoption. Marre et al. (2015) investigated how and to what extent coastal zone managers in Australia used economic valuation of coastal and marine ecosystem services to inform decision making. Over 400 coastal zone managers in different regional, State and Federal contexts of Australia were asked to participate in the survey; 88 survey responses were collected. Results showed that ESV is being used in coastal and marine management in Australia, mostly as a way to communicate and raise awareness, and as a way to support evaluation and discussion during decision-making processes. However, the authors caution that this finding differs across management contexts, such that ecosystem services valuation was reported as most frequently used in commercial fishery use decisions and almost never used in Indigenous and customary use issues.

Baker and Ruting (2014) review the main economic valuation methods within the Australian context, and provide decision trees to help decision-makers decide when investment in non-market valuation may be warranted and to assist practitioners in selecting appropriate methods. They also identify the barriers to the uptake of economic valuation information to inform environmental policy development in Australia. These barriers include a reluctance to apply a cost–benefit framework, scepticism about stated preference methods, opposition from vested interests, lack of familiarity with the methods among decision makers, and time and cost.

Collie et al. (2013) compared 16 marine spatial plans, such as the GBR Marine Park Zoning Plan, from Europe, North America, China, and Australia against a set of attributes of an idealized marine spatial plan that the authors constructed based on published guidelines and recommendations. Eleven of the plans used DSTs. The authors note that the value of a DST tends to increase with the number of planning objectives and trade-offs, but that the amount of data, technical challenges, and cost of tool implementation also increases. Increased transparency in the data, targets, goals and issues being considered were cited as benefits from DST use.

Cvitanovic et al. (2015) surveyed 78 Australian marine scientists on the barriers to knowledge exchange with environmental decision makers. These researchers find a range of barriers to engagement between scientists and decision makers, including inadequate measures of science impact that do not account for engagement activities, a lack of organisational support for engagement activities, insufficient time to conduct engagement activities in addition to other responsibilities and a lack of funding to support engagement activities.

Impediments to the adoption of DST in resource allocation are not unique to marine management: environmental management more broadly suffers from a lack of evidence base. Despite the benefits of DST, it is often observed that they are underutilised, or not utilised at all, by the intended end users (Nilsson et al. 2008; McIntosh et al. 2011). Several reasons are cited in the literature, including: different timeframes between policy decision making and scientific research (Briggs 2006; Cvitanovic et al. 2015); research results not providing the specific information needed to support policy (Pannell and Roberts 2009; Addison et al. 2013); lack of trust in the researchers by policy makers (Gibbons et al. 2008; McIntosh et al. 2011); low capacity of policy makers to use the research outputs in decision making (Rogers et al. 2015); and the lack of a champion within the policy organisation to enable uptake of the research results (Mumford and Harvey 2014).

4. STRENGTHS AND WEAKNESSES OF ALTERNATIVE APPROACHES

The tools described in Section 2 of this report vary in the kinds of problems they address, the resources and technical competencies required to deploy them, their repeatability and transparency, the temporal scales and uncertainties they can accommodate, and the extent to which they may demonstrate diligence in carrying out statutory responsibilities. In Tables 2 – 6 below we coarsely summarise our views on the merit of these approaches in the context of assumed requirements pertaining to logic, data demands, analytical costs (including the costs of acquiring external expertise), openness to evaluation, and the extent to which outcomes are conducive to learning, by decision type.

Table 2 . Acceptable risk decisions.

	Approach		
Criterion	**Unaided judgment**	**Qualitative risk assessment**	**Quantitative risk assessment**
Logically sound	no	partially	yes
Information demands	small	modest	substantial
Cost of analysis	negligible	modest	large
Open to evaluation	no	yes	yes
Conducive to learning	no	partially	yes

Table 3. Summary of common tools and applications for marine park managers dealing with acceptable risk problems.

Tool	**Main strength(s)**	**Main weakness(es)**	**Example application**
Qualitative risk assessment	Ease of use.	Language based ambiguities that invite arbitrary error in assessments.	Management priorities for marine parks (Carey et al. 2007).
Quantitative risk assessment - Logic trees	Simple, visually accessible models.	Can become messy when used for complex problems.	Conservation status of ecosystems (Keith et al. 2013).
Quantitative risk assessment - Bayesian Belief Networks	Accounting for uncertainty and conditional relationships between system variables.	Large requirements for data and/or expert judgment.	Assessment of marine offsets (Jennings et al. 2015).
Quantitative risk assessment – Monte Carlo simulation	Accounting for uncertainty and change over time.	Large requirements for data and/or expert judgment.	Ecosystem modelling (Fulton et al. 2014).

Table 4. Allocation decisions involving management resources.

	Approach				
Criterion	**Unaided judgment**	**Cost effectiveness**	**Benefit cost analysis**	**Programming**	**MCDA**
Logically sound	no	yes	Yes	yes	yes
Information demands	small	medium	Large	large	large
Cost of analysis	negligible	modest	Large	large	large
Open to evaluation	no	yes	yes	yes	yes
Conducive to learning	no	yes	yes	yes	yes

Table 5. Summary of common tools and applications for marine park managers dealing with the capture of preferences.

Tool	**Main strength(s)**	**Main weakness(es)**	**Example application**
Market values approaches	Best approach when markets are open.	Market distortions can lead to bias.	Habitat productivity (McArthur and Borland 2006).
Revealed preference approaches	Able to use prices to estimate some non-market values.	Requires strong analytical skills.	Valuation of dive-based tourism (Pascoe et al. 2014b).
Stated preference approaches	Best approach for non-market values for which revealed preference techniques are unavailable.	Requires sound survey design and strong analytical skills. Cost of administrating survey.	Community valuation of conservation assets of Ningaloo (Rogers 2013).
Benefit transfer	Cheap.	Poor translation from previous studies to current context.	Value of beaches to tourism (Raybould and Lazarow 2009)
Deliberative and other non-monetary valuation	Stakeholder engagement.	Stakeholder preferences may not reflect broader societal preferences.	Prioritisation of marine values (Ogier and McLeod (2013)

Table 6. Summary of common tools and applications for marine park managers dealing with resource allocation problems.

Tool	**Main strength(s)**	**Main weakness(es)**	**Example application**
Cost-effectiveness analysis	Simple to use.	Cannot directly inform circumstances where *status quo* or do-nothing arrangements are best.	Evaluation of alternative control measures for oil spills (Vanem et al. 2008).
Benefit-cost analysis	Most rigorous approach.	Typically requires strong analytical skills and considerable time and resources.	Evaluation of marine protected areas (Rees et al. 2013).
Multi-criteria analysis	Stakeholder engagement	Stakeholder preferences may not reflect broader societal preferences.	Fisheries management (Pascoe et al. 2013).
Programming and optimisation	Able to explore vast numbers of alternatives	Constraints can make trade-offs difficult or opaque.	Fisheries management (Dowling et al. 2011).

Unstructured decision-making is clearly deficient. Although a raft of qualitative and quantitative aspects based on expert opinion can be informally considered in unaided judgment there is no coherent or transparent basis for ordering and aggregating those judgments. In the absence of any structured attempt to disentangle cause-and-effect judgments from value judgments, decision-makers and stakeholders are prey to bias and internal inconsistencies. Environmental disasters are low-likelihood high-consequence events that inevitably invoke regret and remorse. These circumstances lead to availability bias, where the dread of recent events take an elevated position in our collective conscience, and confirmation bias, where stakeholders take an affective, emotion-based position on a desirable course of action and sub-consciously ‘cherry-pick’ evidence that confirms that position, ignoring contrary evidence (Slovic et al. 2004, Hammond et al. 2006). Where adverse events are absent over time, there is a tendency to drift towards a perpetuation of the status-quo or the withdrawal of resources aimed at mitigating risk.

The various techniques available for single-attribute risk analysis do not accommodate exploration of trade-offs among multiple objectives but may be entirely appropriate in regulatory settings where clear (single-attribute) thresholds for acceptable risk are specified. For any single objective, quantitative risk analysis techniques (e.g. Bayes nets, logic trees, Monte Carlo simulation) provide transparent and structured judgments of cause and effect, offering insulation against cognitive difficulties encountered in probabilistic reasoning (Hastie and Dawes 2010). Qualitative approaches may be sufficient in low stakes settings where cumulative and chronic risks are of marginal relevance.

Of the family of techniques that directly address multi-objective resource allocation problems, the simplest is preparation of a consequence table describing alternatives and their pay-offs across the multiple considerations considered relevant. While an effective buffer against poor decisions, a consequence table does not formally capture preferences or score or rank alternatives. At the other end of the spectrum are formal programming techniques, MCDA and benefit-cost analysis. The methodological details and conventions of these approaches are well founded in operations research and welfare economics. But where applied thoughtfully, they often involve significant data demands and relatively high analytical costs. Data demands may include empirical observation or the formal elicitation of expert judgment (Hemming et al. 2018).Even where data are available, the treatment of trade-offs through constraints is a weakness of programming, although this shortcoming may be justified if there is a vast number of potential alternatives that need to be explored (e.g. delineation and zoning of parks).

The costs in conducting non-market valuation studies for benefit-cost analysis to estimate (and monetise) social preferences are often perceived to be prohibitive. This perception may be inflated by a general reluctance to embrace rigour in the assessment of environmental policies, as opposed to say public investment in infrastructure or health. Deliberative approaches built around MCDA and an emphasis on stakeholder engagement tend to be preferred. Implicit in deliberative approaches is an assumption that the trade-offs a subset of stakeholders make in arriving at a decision are acceptable to broader society. In a recent review, Estévez and Gelcich (2015) found that 31 of 95 peer-reviewed publications in marine management and conservation incorporated stakeholder engagement at one or more stages of the MCDA process. However, in general participation was fragmentary. These authors urge greater rigor in promoting an active participation throughout the entire decision process, including value judgments and risk attitudes in the face of uncertainty.

4.1 Tentative recommendations

Providing unambiguous guidance on what DST to use in what circumstances is difficult (but see Schwartz *et al.* 2018 and Bower *et al.* 2018 for recent attempts). In general, greater rigour and detail is warranted as the stakes involved in any decision grow (Dunstan et al. 2015). Beyond this self-evident principle, the authors of this report are not placed to make definitive recommendations on the selection of DSTs in any particular decision-making setting faced by Parks Australia. Here we provide tentative advice which needs to be considered against the time and personnel constraints of the organisation, together with its perceived mandate to make judgments on behalf of stakeholders and broader society.

This report was motivated in part by difficulties encountered by Parks Australia in judging the appropriateness of various approaches to decision-support. This judgment is especially difficult for the many candidate tools and approaches available for resource allocation problems. The logic tree in Figure 17 offers coarse guidance on the circumstances in which different tools and approaches may be applicable. The tree is structured in three sections, the first of which deals with *net public benefit*, the second with who to consult in the elicitation of *preferences*, and the third with *tool selection*.

The first node asks whether or not the level of public investment is a major concern. For example, conservation agencies may hold the view that Treasury does not fund its operations at a level commensurate with society's willingness to pay for the values it protects and the services it provides. The tree emphasises that the only approach for discerning whether or not there is too little (or too much) public investment is benefit-cost analysis underpinned by non-market valuation. That is, BCA is the only tool that can clearly identify circumstances where (a) the level of resources available for allocation is inconsistent with public values, or (b) do nothing is the best alternative because all other candidate solutions have costs that outweigh benefits.

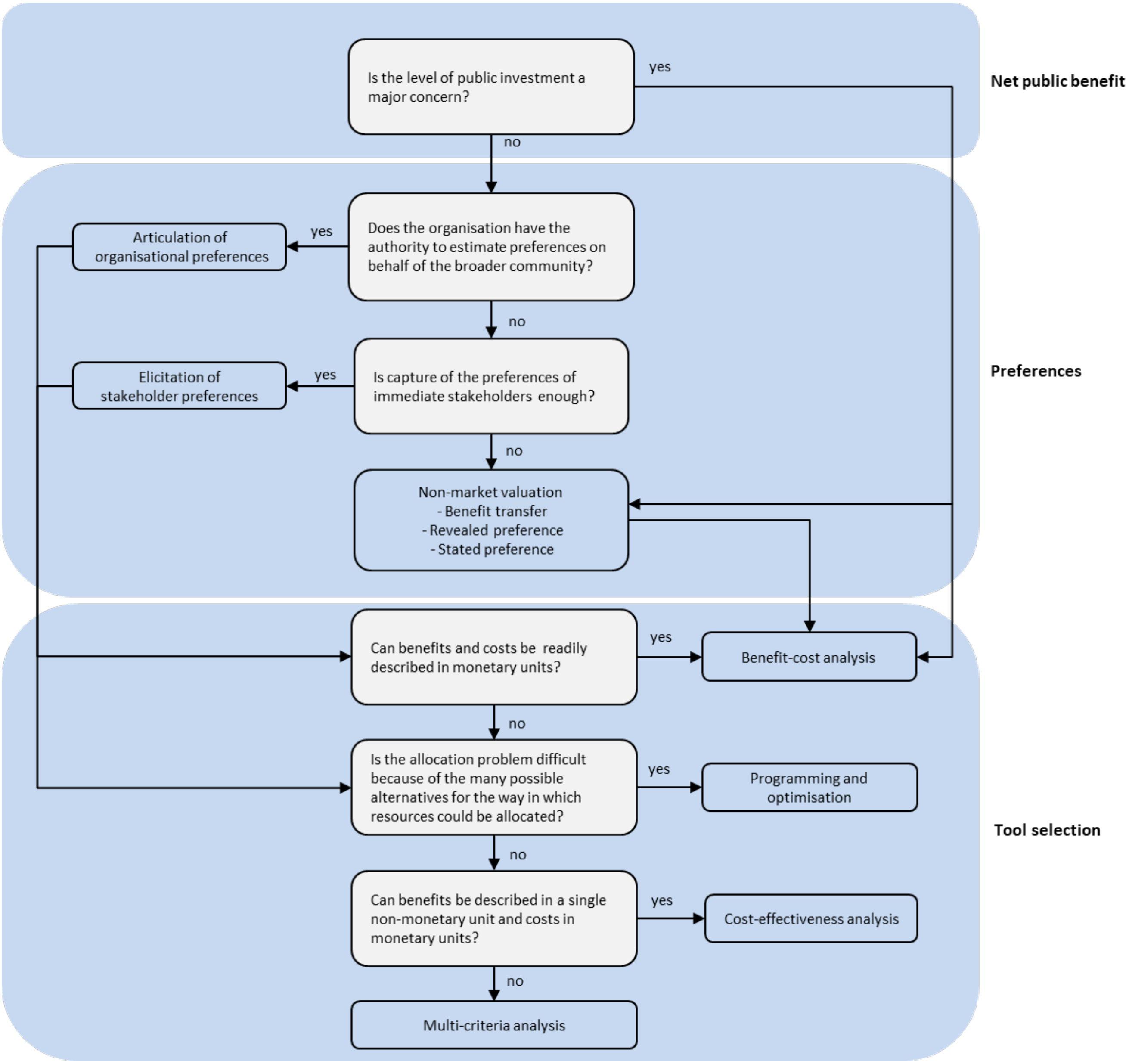


Figure 17. Logic tree to assist in the selection of an approach to resource allocation problems involving non-market values. The chosen path will vary from one decision context to another.

For any specific decision context, the next node asks whether or not Parks Australia regards itself as having the authority to make trade-offs or value judgments on behalf of stakeholders and broader society. In general, the answer will be 'yes' for routine small-stakes decisions and 'no' where the costs of poor decisions will be large or where there is intense stakeholder conflict. Other circumstances in which BCA may be preferentially deployed include those where the values and trade-offs of the organisation or a sub-set of stakeholders may be inconsistent with the preferences of broader society, and where all relevant benefits and costs can be readily described in monetary units. The remaining nodes differentiate circumstances where programming, cost-effectiveness analysis and multi-criteria analysis might be most applicable according to the nature of the problem at hand. We emphasise that there is much room for nuance in the application and interpretation of this logic tree, including the complimentary insights that may be obtained through deployment of the many tools and hybrid approaches described in section 2 of this report.

We note that a potential source of confusion in deciding on an approach to decision support arises from the tendency of scientists to advocate multiple and indirect descriptors of consequences. For example, Failing and Gregory (2003) list the myriad biodiversity indicators they encountered in a resource allocation problem involving an energy development project in a forested catchment:

1. area of mixed wood forest of natural origin;
2. area of deciduous-dominated forest of natural origin;
3. area of wetland;
4. area of non-forest vegetation of natural origin;
5. area of forest of anthropogenic origin (e.g. forestry cut block);
6. area of non-forest of anthropogenic origin (e.g. reclaimed land);
7. density of linear developments (km/km2);
8. density of Crossings of linear developments and watercourses (#/km);
9. number of large forest patches of natural origin (by classes, above);
10. variability in size of forest patches of natural origin;
11. average distance among large forest patches;
12. average edge: area ratio of forest patches;
13. rate of disturbance by wildfire;
14. area by ecosite phase;
15. area of forest by age class (young, mature, old);
16. area of wetland exhibiting patterned fen formation;
17. area of forest with tall trees (.20 m);
18. area of forest with high down deadwood volume (.100 m3/ha);
19. area likely to contain many vascular plants;
20. area likely to contain rare vascular plants;
21. area likely to contain exotic vascular plants;
22. area likely to contain many bird species;
23. area likely to contain rare bird species;
24. number of patches likely to contain rare vascular plants;
25. number of patches likely to contain rare birds.

Failing and Gregory (2003) considered it a ‘mind-numbing task’ to think about evaluating management policies using all of these as valued components of biodiversity. Where a multiplicity of indirect descriptors is used there may be a tendency for over-use and misuse of multi-criteria analysis. An emphasis on outcomes-based management underlines the importance of direct measures of risk and benefit.

For acceptable risk problems, the immediate hurdle to more effective decision-making may be less about the choice of risk assessment methodology and more about defining what is the threshold for acceptable risk for different assets in different places In fisheries management, we note the success the Commonwealth Fisheries Harvest Strategy (DAFF 2007) has achieved via unambiguous guidance on acceptable risk through specification of a clear threshold for a descriptor of direct management interest (biomass in this case) beyond which management intervention is required. There is no technical reason why similar clarity (and success) cannot be achieved in conservation management, so long as the approach adopted enjoys the support of key stakeholders.

In closing, we note that political acceptability is also an important consideration guiding the choice of decision support tools. In the long run, political acceptability is built on trust (Table

5). Many of the more rigorous approaches to decision-support described in this report lend themselves to trust-building through improved competence, objectivity and consistency. Progressing perceptions of fairness may be more difficult to achieve, especially in the context of competing organisational objectives around conservation and sustainable use.

Table 7. Components of trust. (Source: Renn and Levine 1991).

Component	Description
Perceived competence	degree of technical expertise in meeting institutional mandate
Objectivity	lack of biases in information and performance as perceived by others
Fairness	acknowledgment and adequate representation of all relevant points of view
Consistency	predictability of arguments and behaviour based on past experience and previous communication efforts
Sincerity	Honesty and openness
Faith	Perception of 'good will' in performance and communication

A key insight from decision science is how different kinds of trade-offs lend themselves differentially to an erosion of perceptions of fairness and trust. Tetlock et al. (2000) provide a typology of value judgments that includes routine, tragic and taboo trade-offs. Routine trade-offs involve two secular values, for example, fuel efficiency and comfort in the purchase of a car. Routine trade-offs can be cognitively challenging, but their emotional demands are modest. Tragic trade-offs involve two sacred values. The judgment of the extent to which an advantage of specified magnitude for a threatened ecological community compensates for a disadvantage to culturally important sites (or vice-versa) is an example. Taboo trade-offs are those involving a sacred vale and a secular value. The monetary savings (or outlays) required to compensate specified disadvantage (or advantage) to culturally important sites and threatened species are examples.

The emotional demands of tragic and taboo trade-offs lead to avoidance behaviour (Payne et al. 1993, Luce 1998, Hanselmann and Tanner 2008). Alongside flat out refusal to participate, avoidance strategies include 'buck-passing, procrastination, and obfuscation to escape responsibility for making trade-offs that inevitably leave some constituency feeling it has gotten the short end of the trade-off stick' (Tetlock 2000). In the context of AMPs, we contend that although tragic trade-offs between conservation values and cultural values are difficult, they are familiar enough in mature settings where shared responsibility and contrasting priorities are recognised and respected. The more substantial obstacle to sound decision-making may be taboo trade-offs between monetary costs and sacred values. A common avoidance strategy is to resist a considered response to the question of willingness to pay (or accept) and instead provide loose judgments that deny the realities of constrained resources (Tetlock 2000).

Building capacity in decision support is a long term undertaking for any organization (Spetzler *et al.* 2016). But in the immediate future, we suggest Parks Australia might usefully concentrate the development of in-house competencies in (a) analyses underpinning routine decisions for which the organization has clear authority, and (b) accessing appropriate expertise for more challenging decisions, especially those that may compromise standing and trust because of the need to confront tragic or taboo trade-offs.

ACKNOWLEDGEMENTS

The content of this report was developed and improved through dialogue with a range of Parks Australia staff. In particular, we thank Amanda Richley and Dave Logan for thoughtful feedback on earlier drafts. We are also grateful for helpful comment and support provided by the Marine Biodiversity Hub's Nic Bax and Paul Hedge.

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