The motivation for this chapter stems from two premises. First, the field of systems thinking (especially soft and critical) has the potential to improve our capacity to understand and manage the human (i.e. cognitive, social, cultural, and political) elements in wicked water and groundwater issues. Systems thinking approaches provide a holistic framework (of theories, methods, and tools) that can help unlock key management issues and interrelationships from multiple perspectives. In systems thinking, the key assumption is that bringing together different views may lead to building shared, multi-dimensional, and rich understanding of the situation, which may therefore, lead to developing sustainable (i.e. economically-viable and socially-accepted) policies. This assumption presents an opportunity for groundwater policies that cry out for better and more explicit ways of incorporating and linking the human aspects to the groundwater conditions (Richardson et al. 2011).

The second premise motivating this chapter is the perceived lack of studies that explicitly address the systems applications in sustainability problems (including groundwater). Midgley and Reynolds (2004) argue that for every paper on environment management that is explicit about the use of Operations Research/Systems Thinking (OR/ST) methods, there are "at least" five that use similar OR/ST methods, claiming "methodological innovation" without referencing OR/ST. The ideas are clearly useful. Even for authors who mention a systems approach as a research framework (e.g. Bosch et al. 2007), the discussion is often limited to the conceptual idea of systems thinking without explicitly explaining the in-depth implementation details, and how the work links to existing theories and methodologies. Therefore, we aim to promote use of systems methods in groundwater management explicitly rather than reinventing the wheel, and with rigorous reference to theory rather than referring to vague concepts.

The chapter is structured as follows. First, we trace the evolution of the systems idea through the hard, soft, and critical developments (Sect. 24.3). Next, we discuss two important topics in systems approaches: multi-methods and evaluation (Sect. 24.4). In Sect. 24.5, we complement the theoretical overview with a set of selected case studies to shed some light on different implementations of systems thinking and their relevance to groundwater research and management. Finally, we wrap up by drawing some lessons from systems thinking literature and case studies.

## 24.2 The "Systems" Idea

The systems idea is not new, but can be originally traced back to Aristotle's dictum that "the whole is greater than sum of the parts." The contemporary notion of systems can be found in General Systems Theory (GST) which recognizes the importance of interactions and organization (Von Bertalanffy 1950). Since the formulation of GST, the systems idea has developed in two main directions. The first applied the systems idea in biology and ecology. The second resulted in the development of problem solving methodologies, which have evolved through three

**Table 24.1** Summary of the three waves of development in systems thinking approaches

| Point of comparison | Hard | Soft | Critical |
|---|---|---|---|
| System | A system is a well-defined entity that has clear function | A system is a cognitive and social construct that is not independent from the observer | A system is defined by a boundary that may be alienating individuals or a particular social group |
| Purpose | Predicting, optimising, and controlling outcomes | Develop a meaningful understanding, learning | Empowering stakeholders to overcome power imbalances and social inequities |
| Researcher | Outside observer | Participant | Participant, and sometimes, enabler for change |
| Models | Accurate representation of the real world system | Interpretations or intellectual construct to inform debate and learning about possible changes | Interpretations that are used to surface and question assumptions about values, power, and knowledge during a public or corporate dialogue |

waves of thinking: hard, soft, and critical (see Table 24.1). We give an overview of these developments in the following sub-sections.

### 24.2.1 Hard Systems Approaches

During and after World War II, hard systems thinking approaches (optimisation, simulation, systems engineering, systems analysis) appeared as powerful analytical methodologies for solving real world problems. Hard system approaches have been long and widely used to analyse groundwater problems, such as Ayvaz and Elçi (2014). Hard approaches share the following assumptions (Checkland 1981):

- There is a "problem" that can be exhaustively formulated in terms of well-defined objectives and actions that can be optimised or (at least) improved.
- Success in applications depends on quantification of variables and the creation of mathematical formulations that specify the relationships between variables.
- Our knowledge (including models) and language perfectly describe the real world.
- Systems have objective boundaries which are "given" by the structure of reality (Checkland 1983).
- Stakeholders are passive entities who share common views, values and objectives (Rosenhead and Mingers 2001).
- An analyst is an independent observer who perceives the world as a set of interacting components or sub-systems. The analyst's efforts are directed in a *systematic* search for the most efficient means to achieve objectives.

In the early 1970s to the mid 1980s, hard systems approaches came under a lot of criticism for their inability to deal with problems that arise in contexts that are highly complex and involve multiple stakeholders (Dando and Bennett 1981). Many authors, such as Churchman (1970a), reject the view that a system has an objective boundary that is independent of human perspectives. System boundaries are social and subjective constructs. Thus, setting a system's boundary is a critical choice about what the study considers to be relevant knowledge and legitimate decision makers. To account for multiple perspectives, the analysis boundary should be pushed out by "sweeping in" divergent views (Churchman 1970b).

### 24.2.2 Soft Systems Approaches

In response to the attack on hard approaches, soft systems thinking appeared as an alternative approach capable of addressing complex and unstructured situations as it places human and social considerations at the core of systems management. The fundamental distinction between hard and soft systems thinking lies in the way they address the philosophical question about the nature of reality and the nature of knowledge (Checkland 1999). In soft systems thinking, the word systemic is no longer applied to the world but to the inquiry process to explore this world. This view implies that systems thinking remains only as a way of describing knowledge about the world rather than an objective reality (Checkland 1983). For more details, we refer the reader to Mingers (2003) who presented and compared the philosophical and methodological assumptions underpinning a multitude of hard and soft systems approaches.

In the soft view, it is necessary to engage in an iterative process of *systemic* inquiry and learning (which may be ongoing) to develop a *meaningful understanding* of the situation (Checkland 1985). To build a meaningful understanding, we need to understand the cultural and social aspects of the situation, as well as the purpose, interpretations, and actions of stakeholders. This includes all people who may affect or be affected by the action outcomes, including the researcher(s). The soft approach takes the view that scientists bring their own values and subjective interpretations with the aim of intervening within the system. Intervention is defined as "purposeful action by an agent to make change" (Midgley 2000). Active stakeholder engagement and an active researcher role are common pillars of soft systems and also action research. For a detailed discussion about the links between systems approaches and action research, readers are referred to Flood (2010).

Within the soft systems approach, several methodologies, known as Problem Structuring Methodologies (PSM), have been developed in the literature, including:

- Soft Systems Methodology (SSM) (Checkland 2001)
- Cognitive Mapping (CM) for Strategic Options Development and Analysis (SODA) (Eden and Ackermann 1998)
- Viable System Model (Beer 1989)
- Visioning choice methodology (O'Brien and Meadows 2007)

Whereas PSMs have different forms, they share four generic phases (Mingers 2000):

1. Appreciation of the situation as perceived by stakeholder groups
2. Analysis of the structure that generates the perceived situation
3. Assessment of ways of changing the situation into more desirable conditions
4. Action to implement change, and achieve desired outcomes.

PSMs use models/modelling in a heuristic fashion, as learning aids or artefacts that help system stakeholders to co-construct a meaningful understanding, but are never taken to represent reality.

### 24.2.3 Critical Systems Thinking

Soft systems thinking and PSMs have been criticized for not being able to address the question of power relations, how they influence the problem situation, and how they are perceived by system actors. This has resulted in a third wave of systems approaches: Critical Systems Thinking (CST) (Ulrich 2000; Jackson 2006). CST rests on the key notion of "boundary judgment", and how it determines how people perceive and judge a particular situation (in relation to what is and what ought to be the case) (Midgley 2000). Setting a system boundary is a critical choice about what the study considers to be relevant knowledge and legitimate decision makers. Ulrich (1994) argues that exploring boundaries through dialogue among stakeholders make the analysis more "rational" and robust than an external group of experts (e.g. scientist and policy makers) imposing their own values. Midgley (2000) argues that boundary setting is deeply underpinned by a value judgment; and conflict arises when two or more value/ethical systems come into tension. Boundary selection therefore has ethical implications (Midgley 1992). People draw a boundary around issues they perceive as sacred or central. Issues outside their boundary are regarded as subsidiary, which marginalises people that hold that view. From this perspective, CST aims to explore and make explicit different boundary judgments, and help justify why a particular boundary judgment is selected. It proposes a dialogical framework to allow for collective reflection to acknowledge and negotiate sources of motivation, power, knowledge and legitimation.

Inspired by the critical systems idea, several methodologies have been developed in the literature including:

- Critical systems heuristics (CSH, (Ulrich 1994))
- System of systems methodologies (Jackson 1999)
- Systemic intervention (Midgley 2000)

## 24.3 Multi-Method and Evaluation in Systems Approaches

The design and implementation of systems thinking interventions depends on choosing effective methods and allowing for an adaptive process. In this section, we discuss two key topics to achieve this aim: use of a multi-method approach and evaluating systemic interventions.

### 24.3.1 Use of a Multi-Method Approach

The debate about the three waves of systems thinking has moved away from arguing the strength of each wave to recognizing that the three waves take different, but not incompatible, perspectives on the world. This view has resulted in the rise of multi-method/multi-methodology as a framework to accommodate different views of systems (Mingers and Leroy 2010). In the context of this chapter, we will use the term multi-method to denote the broad idea of combining methodologies and/or methods (i.e. hard, soft, and critical) within a real-world intervention (Mingers 2000). Multi-method is increasingly regarded as an essential framework for dealing with wicked and turbulent environments (Mingers 1997).

Wicked problems have multiple dimensions: physical or material, personal and social dimensions. Multi-method strengthens the inquiry process and provides multiple lenses for exploring different aspects of multi-dimensional situations. Whereas the intervention process passes through a number of phases, some methods however can be more useful than others for different phases. Pulling the two ideas together, Mingers and Brocklesby (1997) developed a framework to map out how systems methods can be used to examine the problem dimensions across the different intervention phases. Several research directions have stemmed from the multi-method idea, such as: "coherent pluralism" (Jackson 1999), "pragmatic pluralism" (White and Taket 1997), and "creative design of methods" (Midgley 1990). Whereas these approaches share the idea of using multi-method, they have different view about how methods are selected and employed. Kotiadis and Mingers (2006) identified two strands in multi-method research: (1) those who think that methods and methodologies can be effectively mixed-and-matched to strengthen the inquiry (sometimes referred to as pragmatists); and (2) others who are concerned about the incommensurability of paradigms, and accept multi-method legitimacy only under the condition that it respects the theoretical underpinning of the process used to combine methods. For an overview about the evolution of multi-method theory, readers are referred to Zhu (2011). In practice, there have been different forms of applying multi-methods, such as using methods in parallel (e.g. use of two problem structuring methods at the same time to inform each other) or in series (e.g. use of outputs from problem structuring to inform the design of a numerical model).

### 24.3.2 Evaluating Systemic Interventions

The topic of evaluating systemic methods has gained increasing attention in the literature (Midgley et al. 2013). A similar trend is observed in environmental studies (e.g. Matthews et al. 2011; Bellamy et al. 2001). Several reviews on systemic research have concluded that although projects claim to have achieved some kind of value, the evidence is merely based on the author's own reflection with minimal formal evaluation (Midgley 2007). Howick and Ackermann (2011) conducted a comprehensive review of multi-method systems applications, and concluded there is often a limited link between a project's rationale, process, and actual outcomes.

Similar to the debate about hard and soft system approaches, there exist two main positions for evaluating systemic interventions (White 2006): Positivist and interpretive positions. Positivist evaluation aims to collect *objective* data about the efficiency and effectiveness of the methods in practice. This can lead to "universal" evaluations that may applicable across multiple interventions (Rowe and Frewer 2004). On the other side, an interpretive evaluation approach argues that objective evaluation has limited practicality and relevance to gaining insights into worldviews and interpretations. Instead, it frames evaluation as a continual learning process about the methods and how they have been applied in reality (i.e. what worked, what did not work, why). Along the same lines, Checkland and Holwell (1998) argue that recoverability rather than repeatability (of process and results) is to be used as the criterion for evaluating systemic research. For 'recoverability' to be achieved, the whole research activity or intervention, including the methodology to be employed must be made explicit for an outsider.

Recent developments have argued that there is a need for new evaluation theories or frameworks that combine both positivist and interpretive positions into the systemic intervention, such as (Midgley 2007). Towards this goal, a few evaluation frameworks have been developed, such as White's pragmatic theory-based framework (White 2006). Whereas systemic evaluation frameworks have different forms, they can share some common ideas or principles:

- The need to focus on the *purpose* of the evaluation and how it is meaningful and relevant to the intervention's purpose and participants' worldviews
- The researcher has to be pragmatic about what they can and cannot measure in complex, contested and resource-limited contexts
- Both quantitative information and qualitative insights are essential and complementary means of establishing evidence
- Use of established theories (e.g. a behavioural theory) to support the design of evaluation and/or explain results add rigor and enrich findings
- The evaluative inquiry itself is a systemic and learning process that involves multiple perspectives (e.g. who and what determines successful outcomes). The process needs to involve continuous exploration of the: purpose, context, and methods and how they link together.

## 24.4 Systems Approaches in Practice and Learning Lessons

So far, we have given a theoretical overview of systems approaches. Now, we complement the picture by giving the reader a feel for applications of systems approaches in natural resource management in general because of the very lack of groundwater-specific studies. Paucar-Caceres and Espinosa (2011) surveyed systems applications in environment and sustainability areas published in leading OR/MS journals, and concluded that the majority of applications belong to the hard view, with very few soft and critical approaches.

In this section, we present a selection of case studies. We aim to shed some light on elements in the context of each case study (i.e. purpose and methods) that can be relevant in groundwater systems. We hope that this may encourage the reader to think whether and how they can make use of the systems approach in their groundwater applications. We selected case studies to represent different *forms* and *purposes* of systems methodologies (See Table 24.2) where:

- Form: single and multi-method interventions where multi-method cases present different combinations of methods (hard, soft, and critical)
- Purpose: intervention's aim is to develop an end-product (e.g. decision support tool), or a process (e.g. a medium for learning and exchanging views)

### 24.4.1 Support Community Engagement in Water Conservation Policies in New Zealand (Foote et al. 2006)

#### 24.4.1.1 Context, Purpose, and Design

The study takes place in a water-stressed town in New Zealand where water security stands as a contentious issue between government agencies and the local community as a result of the failure of successive policies to provide a satisfactory solution to ongoing water shortages. The study starts with the premise that effective implementation of water conservation policies depends on the collaboration of all stakeholders and interest groups. The purpose of the study is to provide a legitimate participatory process for engaging stakeholder groups in evaluating water conservation policies employed. Legitimacy is sought by involving independent third party scientists who are trusted to bring different views to the negotiation table.

**Table 24.2** Summary of case studies reviewed as part of this chapter

| | (Foote et al. 2006) | (ElSawah 2010) | (Powell and Osbeck 2010) | (Larsen 2011) |
|---|---|---|---|---|
| Form | Multi-method (boundary critique + rich picture) | Multi-method (system dynamics + cognitive mapping) | Soft systems methodology | Critical systems heuristics |
| Purpose | Process-driven | Product-driven | Product-driven | Process-driven |

In response to the perceived tension, authors foresee the potential of using a boundary critique (Midgley 2000) method to: (1) make the problem definition explicit from a variety of viewpoints; (2) identify areas of agreement, disagreement, marginalisation, and sources of conflicts among stakeholder groups; and (3) guide how problem structuring methods are selected, used and mixed in the case study, and identify implications for inclusion, exclusion, and marginalisation of issues and stakeholders. Authors used interviews, rich pictures and scenario planning workshops to identify and share a multi-perspective evaluation of water policies. Results from using boundary critique show that the conflict about the effectiveness of water conservation measures is rooted in the tension between the pro-development and anti-development values of citizens, and that the debates about water policies cannot be "decoupled" from the "wider debate on the desirability of economic development."

Later in the process, authors reported facing an "ethical dilemma" about the credibility of their research given the decision of policy officers to overlook issues raised by the community (i.e. economic development impacts on water security), and their unwillingness to discuss all the information identified through the engagement process. However, authors decided to think strategically by keeping the process alive, building strategic relationships with key parties, and exploring potential opportunities to establish more open dialogue in the future.

#### 24.4.1.2 Evaluation

Reflecting on the methods used, authors noted that systemic intervention can establish a "sound process" of stakeholder engagement, but it does not necessarily guarantee "win-win outcomes for all." They found that boundary critique provides a useful way for exploring values and boundaries. However, results should not be taken for granted in that there are limits to the method's capacity to elicit values and boundaries.

Whereas the paper is transparent about the methods used, it does not justify how the boundary critique method informed the choice to use rich pictures as problem structuring, although the justification of how methods were selected was as an explicit objective at the outset.

Reflecting on the process outcomes, authors perceived the project as a success because it improved understanding and dialogue as expressed by participating groups. Quotes are used to establish evidence, for example: "*The decision-making tools. . .have allowed a wide range of stakeholders to be actively involved in the decision-making process. The methods employed have provided a non-threatening environment for stakeholders to express their views and this participation has led to general acceptance of the consultation outcomes*." In addition, the evidence that the client invited the authors to do more work in the area supported the project's success.

#### 24.4.1.3 Relevance to Groundwater Research and Management

This work has a strong relevance for community engagement in groundwater planning. For example, the concept of acceptable or sustainable aquifer yield is underpinned by a judgment about the spatial boundaries (e.g. geographic area, aquifer), temporal boundaries (e.g. planning cycle), administrative and institutional boundaries (e.g. government levels), value boundaries (e.g. social, economic, and environmental groups), and knowledge boundaries (e.g. scientific vs. local). Who makes these judgments? How are these judgments made? Are judgments and their implications transparent to all stakeholders or hidden and scattered across the governance system? In a sound and legitimate planning process, these questions need to be identified and negotiated among stakeholder and interest groups. Boundary critique can be a useful method to facilitate these discussions.

### 24.4.2 Communicating About Water Security Issues in the Australian Capital Territory (ElSawah 2010)

#### 24.4.2.1 Context, Purpose, and Design

The study is based on three premises: (1) people have over-simplified mental models about the causal interactions that drive the behaviour of a water resource; (2) flawed and inaccurate mental models may lead to less informed decisions and attitudes towards water management policies and conservation measures; and (3) the design of effective communication tools needs to be based on sound understanding of such mental models, and best ways to improve them. The purpose of this work was to develop an interactive dynamic simulator that could be used to inform and improve the mental models that water users and managers have about the complexity and uncertainty surrounding the future of water security in the Australian Capital Territory. A cognitive mapping method was used to elicit, analyse, and visualise the mental models of water users and managers, specifically in relation to misperceptions and erroneous assumptions, sources of conflicts and communication gaps.

Although managers frequently point out the need to "get the community on board" and for two-way communication, they were reluctant to engage in open discussion groups, indicating that to do so would be overly confronting and excessively time consuming. Yet, they welcome the use of a model as an online educational tool to improve public understanding about the complexities of water management. Given that the primary purpose of the project was developing a modelling tool, the author had to find other data collection and validation methods (e.g. interviews and electronic data sharing methods) to share results and gain feedback. Based on these data, a series of conceptual and numerical system dynamics models were used to develop an interactive simulator that can be used to check the dynamic coherence of elicited mental models and views. A transparent flow of information from cognitive mapping, to conceptual, and then numerical system dynamics models helps users relate their thinking to the end product, and makes the modeller be explicit about the modelling assumptions.

#### 24.4.2.2 Evaluation

The author uses both self-reflection and pilot experiments to evaluate the modelling process (e.g. transparency, relevance) and its outcomes (e.g. improving the mental models that water users and managers have before and after interacting with the simulator). The author reported that the process allowed for identifying the different perspectives and mental model without prior assumptions. However, the process was limited to only two stakeholder groups, and did not address any of the power relationships in the system and how they may affect policy making. The use of rigorous experiments to evaluate the learning outcomes gives an understanding of what particular perceptions the model can influence.

#### 24.4.2.3 Relevance to Groundwater Research and Management

This work has two key implications for communication and modelling in groundwater management systems. First, the invisible nature of groundwater resources compounded by lack of scientific understanding about the system breeds misconceptions among lay people about the resource's nature, and how it changes (e.g. the myth of underground rivers). Grounded on cognitive psychology, cognitive mapping enables in-depth understanding of these mental models and their implications for attitudes and behaviours.

Secondly, from a modelling viewpoint, modellers often select the boundary of the system to be modelled (what to model) through "ignorance and/or politics" (Eden 1994). The ignorance option is the default for most modellers who decide to ignore the problem complexity and model what they think important to model. Or alternatively, modellers may choose what to model based on how individuals or groups in power (e.g. experts, policy makers and scientists) define the problem. As an alternative, cognitive mapping provides a cognitive approach for modelling where the modeller starts the modelling process by seeking the idiosyncratic views of problem owners. The decision of "what to model" naturally flows from the way problem owners think about the problem. The modelling progression provides better ways of incorporating stakeholder's views and mental models into models.

### 24.4.3 Stakeholder Realities in Mangrove Rehabilitation Processes in Southeast Asia (Powell and Osbeck 2010)

#### 24.4.3.1 Context, Purpose, and Design

The project starts with the premise that "underlying problem definition" significantly affects the design of initiatives to rehabilitate the mangrove forests in East Kalimantan, Indonesian Borneo. Soft systems methodology is used to support the critique of the rehabilitation planning process from the perspective of different stakeholders in the system.

#### 24.4.3.2 Evaluation

The authors did not reflect thoroughly on the choice and use of their method, but the general impression is that the method was successful in eliciting multiple perspectives and understanding the differences between them.

#### 24.4.3.3 Relevance to Groundwater Research and Management

Groundwater planning is often evaluated from a policy compliance perspective, judging its success from a policy maker's viewpoint. The use of SSM may provide multiple lenses for incorporating other views, especially of those who will implement and be affected by the policy. Some of the differences in viewpoint in this mangrove rehabilitation case transfer to a groundwater management context. There is conflict in objectives between scales, particularly national, state, district and individual, and between groups. Each group adjusts their behaviour to cope with this conflict, leading to unintended consequences and failure to meet objectives. This paper concludes: "The owners' worldview has been shaped by the widely accepted regional assumption that there are strong linkages between the ecological services provided through the rehabilitation of mangroves and the livelihoods of local coastal communities contrary to the assumption, the implementation of this worldview has led to a transformation that neither promotes the cause of conservation nor contributes to sustainable livelihoods of local community. Rather, the beneficiaries have been a private elite. The victims have been the most marginalized in the community and ultimately the ecosystem in which these processes are nested." The distributed nature of groundwater pumping and use can lead to similar self-organising behaviour at multiple scales. Understanding the points of view of different groups can help the plan to avoid such failures.

### 24.4.4 Facilitate Stakeholder Dialogue About Coastal Conservation Policies in the Philippines (Larsen 2011)

#### 24.4.4.1 Context, Purpose, and Design

The study takes place in the northern Philippines where there are growing concerns about overfishing, a declining fisheries industry, low community engagement in integrated coastal management, and hidden agendas overshadowing coastal planning. Within the context of existing involvement in stakeholders in planning, the study aims to facilitate dialogue as a process of social learning, to allow sharing of multiple perspectives on defining the problem and its solutions. The ultimate social learning aim is to enable "stakeholder self-organization." The researcher planned to use Critical Systems Heuristics (CSH, (Ulrich 1994)) to facilitate boundary critique and "provide a 'liberating language' for citizens." However, "participants commented about the rigidity [of CSH] and felt constrained by [its] structure". The author decided to use CSH in an exploratory sense with some communicative tools, such as Venn diagrams and mind mapping.

#### 24.4.4.2 Evaluation

The researcher reflected on the process and concluded that the success of any method is highly dependent on context and implementation. The process was modified as it progressed based on feedback from participants. Underlying boundary problems were identified. However, the paper does not say how the process ended.

The research is built on solid theory, states a clear goal to be evaluated and maintains a reflective approach to both. While we do not know the end outcome, this is a good representation of a well-performed system intervention

#### 24.4.4.3 Relevance to Groundwater Research and Management

Fisheries and groundwater resources are both shared resources, resulting in similar problems of degradation of the resource in a 'tragedy of the commons' when individuals do not have strong feedback on the effect of their actions on the resource. In both fisheries and groundwater management, this has commonly been dealt with through "command-and-control management," where individual actions are regulated by law. Opposition to this arrangement has led to "a shift towards increased stakeholder participation." However, these altered arrangements have their own weaknesses. This paper's attempt to support "social learning for self-organisation" can therefore also be useful in groundwater management, to establish new relationships or restructure existing ones to allow people affected by resource degradation or resource management to participate. The paper supports the claim that tools that help stakeholders participate in management must be used within a broader systems approach, allowing the process to evolve as new information is gained. Addressing the complexity of the human dimension cannot be a simple recipe, "a continuous reconstruction of the process and its assumptions was necessary."

## 24.5 Lessons Learnt

In this section, we share some of the lessons for applying systems thinking interventions, which are manifested in the presented case studies.

First, no single discipline can provide all the answers to addressing human aspects of groundwater management. In particular, systems practitioners and researchers should not think or present their methodologies as being the "most effective", or most comprehensive, pluralistic or holistic (otherwise, they would have fallen into the managerialism thinking trap themselves!). Instead, researchers should have a reflective spirit where they fully understand the strengths and limitations of different methods, and communicate openly about implications for the process and its outcomes.

Second, the effectiveness of a method is strongly dependent on purpose, context and implementation. Whereas most (if not all) systems interventions end up developing both processes and products, it is essential for the researcher to have a clear understanding of the primary focus of the intervention (i.e. process-driven or product-driven). This influences process design, including: choices of methods, ways to mix them, evaluation design, as well as strategies to cope with gatekeepers and lack of information.

Third, existing literature on the theory and practice of systems approaches provide rich guidance on how to select, design and implement methods. While practitioners and researchers need to be aware and be explicit about their research's theoretical and methodological stance, they still need to be creative about how they adapt and localise the approach for their case study requirements and constraints.

Finally, incorporating human elements into analysis brings up challenges that often need to be overcome by modifying the existing approach. For example, there may be times where stakeholder groups (as individuals or groups) will act as gatekeepers and try to influence or even block the process and its potential outcomes. It is essential for researchers undertaking this type of research to identify those gatekeepers and develop techniques to work around challenges, such as by looking for other information sources, building trust with key parties, and instituting flexible arrangements to accommodate concerns.

## 24.6 Conclusions

Groundwater management issues present a serious challenge partly because of the complexity and uncertainty that human elements (i.e. cognitive, social, cultural and political) bring into the problem, as well as our limited capacity to fully comprehend and deal with such elements and their interactions with the biophysical systems. Whereas there is a wide recognition of the importance of stakeholder participation for the design and implementation of effective policies, the ongoing depletion of groundwater and disputes surrounding management policies suggest the need for better participatory mechanisms. This raises the question of how human elements can be incorporated into groundwater policies. Whereas there is no single discipline that can provide answers for such crucial research and policy questions, this chapter argues that systems thinking (especially soft and critical approaches) has the potential to provide a framework of theories, methods and example applications to help incorporate human elements into groundwater management and research. This chapter aims to give an overview of systems thinking by firstly describing the theory, distinguishing between hard, soft and critical systems thinking approaches. Secondly, we discussed the importance of mixing methods from these approaches and evaluating 'process' and 'outcomes' when applying them. Thirdly, we reviewed four example applications, and highlighted their relevance to groundwater management systems. Together, these three elements indicate how the framework of systems thinking can help with a number of issues that manifest themselves in groundwater management and research, including: understanding and learning to account for different points of view in planning; understanding how groups affected by a change might respond; helping to enhance participants' view of the problem; exploring conflict; and critiquing existing management and groundwater use arrangements with a view to improving them.



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# Decision Support Systems and Processes for Groundwater

25

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**Abstract**

Information and knowledge management challenges abound in groundwater sciences. Groundwater problems of interest to society are characteristically complex and exceed our ability to solve them without the aid of computational analysis. Yet discipline specific problems that are of interest to hydrogeologists frequently do not directly address the immediate decision making needs of policy makers, groundwater managers, and stakeholders. It is the immediate societal needs that drive the demand for science-based information for common problems in which groundwater figures as a prominent element. Integrated Assessment and Modeling (IAM) presents an approach for merging discipline and case-specific knowledge, such as those in hydrogeological sciences, with social drivers for use in decision support applications. Moreover, decision support systems (DSS) that are constructed and applied using integration as a guiding principle and design ethic can advance groundwater DSS beyond passive support toward active and, eventually, proactive support for implementations to achieve real world integrated groundwater management.

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## 25.1 Introduction

Groundwater is a critical water resource that must be managed effectively while meeting the demands of society. The behavior and response of groundwater systems to natural and human influences are best understood through scientific analyses using data and models. In groundwater resource management, as with all water resources, disputes can be compounded by misconceptions about the meaning of data and scientific models, as well as social and political misunderstandings among the various interests. The complexity of groundwater management creates the need for computational assistance to support reasoned consideration of available scientific knowledge in conjunction with the preferences of the resource users.

Decision support systems (DSS) are computational systems that use data and models interactively to aid in the formulation, analysis, and selection of management strategies. The design, architecture, and implementation of DSS are extensive, highly variable, and, ultimately driven by the needs of the decision problem and instance that is under evaluation. At the simplest levels, DSS may provide repositories of data and information in accessible formats and could offer tools to search and discover repository content. At the other end of the spectrum, DSS may incorporate sophisticated simulations, link with optimization algorithms, or other intelligent systems components to enhance decision making. Regardless of the level of sophistication, DSS are well suited for application to integrated groundwater problems because they can provide a set of applications, methodologies, and tools to cope with the inherent complexity and uncertainty. They can also be part of an Integrated Assessment and Modelling (IAM) process (Jakeman and Letcher 2003) providing distinct advantages for facilitating the IAM process, its transparency and its legacy. Indeed if constructed appropriately, DSS can provide ways of exploring and explaining tradeoffs, provide a tool for adoption and adaptation, create a repository to document the project methods, archive a library of integrated data sets, models, methods, visualization and other tools, a focus for integration across researchers and stakeholders, and act as a training and education tool (Jakeman and Letcher 2003). While the use of DSS for groundwater problems poses potential for improved outcomes, in practice DSS technologies are rarely implemented.

Conceptually, the use and adoption of DSS for groundwater is straightforward. Yet the adoption of DSS may be limited due to scientific, social and technical challenges (McIntosh et al. 2011). Groundwater decision support combines collections of scientific data and models that are inherently uncertain, so that drawing robust recommendations for policy or management is difficult. The creation of DSS is also a multi-disciplinary process that engages subject matter expertise with stakeholder interests across a wide range of sectors in society. Framing DSS applications so that the inputs and outputs are relevant for multiple perspectives is an added hurdle between theory and practice. While the level of effort for developing hybridized computer architectures for DSS is decreasing, the length of time, costs, and computational intensity remain barriers to regular use for groundwater.

This chapter evaluates the state of DSS applications that incorporate groundwater modules with the aim of informing researchers and practitioners interested in designing, developing, and deploying DSS for use in integrated groundwater management.

## 25.2 Decision Support Systems in Relation to Groundwater

Population is increasing around the globe with over 9.5 billion individuals projected by 2050 (United Nations 2010). The concomitant water resource demands for these 9.5 billion water users are expected to lead to disputes over the finite global water supply. To address future water demands, groundwater science needs to provide adequate characterization of the physical systems to assure that policy limits, and management strategies for water allocation are feasible. Simultaneously, scientists and managers need to incorporate the concerns and priorities as defined by stakeholders and the policy context for any aquifer early in design and assessment of options. In effect, knowledge related to both aquifer performance and groundwater governance needs to be explicitly provided in usable formats, such as DSS, in order to achieve integrated groundwater management (Pierce et al. 2013).

Integrated methods that incorporate considerations beyond hydrogeologic analyses using a strict disciplinary focus can be employed to assess the factors of aquifer management or policy defined by both science and consensus conditions (Pierce et al. 2013). The continuum view of aquifer yields (Pierce et al. 2013) fits within an integrated water resources management approach to groundwater science and lends itself to decision support applications. It also requires an adjustment to the underlying framework hydrogeologists use to describe and categorize types of yields. Every DSS is built using datasets and models that represent the problem domain and key elements of interest to decision makers and stakeholders. Building on the concept of inter-related knowledge processes, Fig. 25.1 highlights the relationship between decision

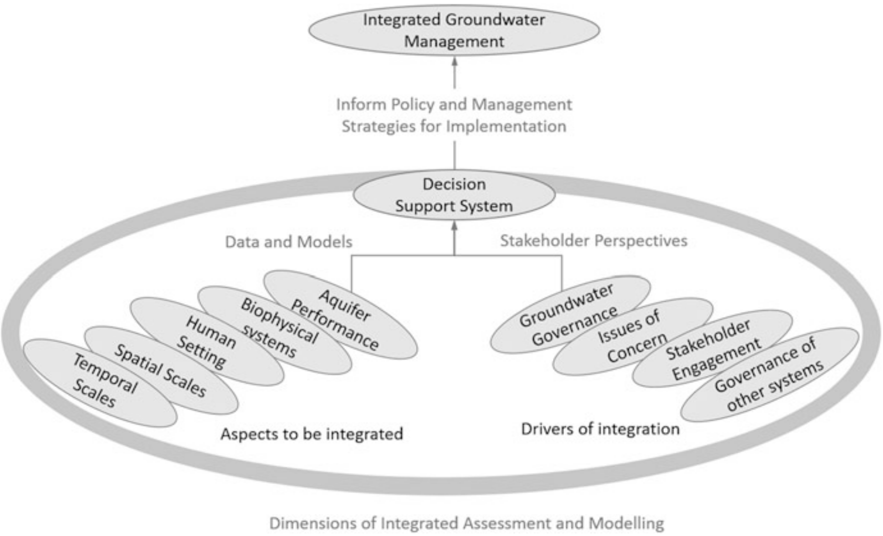


**Fig. 25.1** The conceptual relationship between decision support, aquifer performance, and groundwater governance in integrated groundwater management (Modified from Hamilton et al. 2015; Pierce et al. 2013)

support and the knowledge processes of aquifer performance and groundwater governance. It depicts an expanded scope of DSS for applications in integrated groundwater management by combining framing elements from hydrogeological sciences and an aquifer continuum approach (Pierce et al. 2013) and the primary dimensions of IAM (Chap. 1 and Hamilton et al. 2015).

Beyond the content, disciplinary expertise and relationship among the interacting parts of a DSS process, the type of support can vary from informative to normative. The targeted approach distinguishes between providing access to explanatory or analytical information about a decision problem (informative) versus approaches that provides guidance on candidate solutions (normative). This distinction is a factor in determining the selection, incorporation, and interaction with the scientific information and knowledge that becomes part of the DSS build for each application.

### 25.2.1 Aquifer Performance

Science-based decision making depends upon an acceptable understanding of groundwater systems. Hydrogeology describes aquifers and groundwater flow principally through the use of data and models. Aquifer performance factors reflect physical processes commonly assessed through geological observations, and field measurements of flow conditions that are encoded and integrated into simulation models by subject matter experts (Pierce et al. 2013). Groundwater science has made significant strides towards measuring, describing and quantifying the nature of aquifer behavior. Some traditional hydrogeological methods for measuring or estimating groundwater parameters (see also Chap. 3) include water budgeting, numerical modeling, optimization, simulation, chemical tracing, chemical mixing models, flow-net construction, pump testing, slug testing, and geophysical methods (Weight and Sonderegger 2001).

Field observations and the principles of flow that are used to evaluate groundwater response also provide a set of natural attributes that are common to hydrogeologic problems. Hydrogeologic attributes (shown in Table 25.1, Sect. 25.3.1 of this chapter) are the most basic unit of information for describing groundwater systems. As such, hydrogeologic attributes form the cornerstone elements in an ontology for groundwater decision support. Ontologies are formal representations of knowledge. The set of vocabulary, concepts, and the relationships between them are defined within a domain. In this case, the hydrogeology domain has established a set of information within an ontology to describe how groundwater systems function. A first step towards designing, developing, and using hydrogeological information to support decisions depends on identifying what kind of information and knowledge is necessary to describe the problem adequately. Physical system attributes for groundwater are the first necessary elements. A secondary set of necessary elements includes the considerations related to stakeholder concerns and revolves around the topic of groundwater governance.

**Table 25.1** Natural attributes for a hydrogeologic system[a]

| State conditions | Inflows | Storage | Outflows | Model considerations[f] |
|---|---|---|---|---|
| Aquifer type[b] (m) | Natural recharge spatial component [$R_n(x, y)$] | Specific storage ($S_s$) | Natural discharge spatial component [$Q_n(i,j)$] | Planning Horizon (g) |
| Boundary conditions[c] | Natural recharge rate [$R_n(t)$] | Saturated thickness (b) | Natural discharge rate [$Q_n(t)$] | Stress Period (p) |
| Areal extent of aquifer[d] (A) | Artificial recharge spatial component [$R_a(x, y)$] | Storage ($S_T$) | Pumping well spatial component [$Q_a(i,j)$] | Time Step (t) |
| Porosity (ø) | Artificial recharge rate [$R_a(t)$] | Specific yield ($\phi_{eff}$ or $S_y$) | Pumping well discharge rate [$Q_a(t)$] | Cell (i,j,k,z) |
| Hydraulic conductivity (K) | Return flow ($\alpha$)[e] | Storativity [−] | Evapotranspiration [$Q_e(t)$] | Zone (z) |
| Land Surface Elevation ($m_{ij}$) | Lateral or vertical influx (V)[e] | Hydraulic head [$h(x,y,z)$] | Lateral or vertical outflux (V)[e] | Bottom confining unit elevation ($n_{ij}$) |
| Drain elevation (d) | Unrecoverable Storage ($S_u$) | Minable Storage ($S_m$) | Replenishable Storage ($S_r$) | Diffusivity[g] (T/S) |
| | | Transmissivity (T) | | Acceptable variance (X) |

Notes:

[a]Table excerpted from Pierce 2006 showing a list of influential hydrogeologic parameters as indicated by Feinerman and Knapp 1983; Gisser and Sánchez 1980; Bredehoeft and Young 1970; Freeze and Massmann 1990; Alley et al. 1999; Kresic 1997; Harbaugh and McDonald 1996; Kalf and Wooley 2005 – this list is not necessarily comprehensive

[b]Such as fractured/porous; consolidated/unconsolidated; stratigraphic position and extent (after Freeze and Massmann 1990)

[c]Conditions can include no flow boundaries (lateral), surface impermeabilities, constant heads, differences between geologic units, etc.

[d]An areal extent may be subdivided into zones of confinement, unconfined, and artesian

[e]Return flow and lateral influx or outflux can be counted within the artificial or natural recharge and natural discharge components respectively or split apart as separate components of recharge to the system as shown here

[f]Presented in the context of finite difference modeling, such as in MODFLOW packages

[g]Diffusivity is an indication of the rate of movement through a system and the capacity to sustain localized drawdowns without resulting in long-term storage depletion. An aquifer's diffusivity is probably a good indicator of the relationship to an appropriate planning horizon

### 25.2.2 Groundwater Governance

Management of water resource demands requires the incorporation of legal and regulatory rules for allocation (Part II of this book) as well as community preferences for risk sharing of the potential consequences of water shortages. In short, the interdependency of community drivers and science-based analyses must be recognized and integrated to determine the actual availability of a resource under various management policies as depicted in Fig. 25.1.

Aquifer governance includes the social and contextual aspects of a case that may be used by groundwater managers, together with operational definitions, to implement management regimes (Pierce et al. 2013). Participatory processes are one of many stakeholder engagement and modelling approaches that are well suited for unravelling the issues of aquifer governance. A review of design methodologies, approaches, and guidance on common stakeholder modelling techniques and typologies are discussed broadly in the literature on decision support processes and stakeholder engagement (e.g. Voinov and Bousquet 2010; Margerum 2008). Combining scientific knowledge with stakeholder perspectives, preferences, and concerns generates opportunities to (1) address misconceptions about the science content, (2) establish a shared learning and visioning environment, and (3) increase the likelihood of adoption for solutions that may be identified. DSS offer mechanisms and methods for merging a plurality of views and information that are needed to achieve effective groundwater governance and reduce the potential for conflict.

### 25.2.3 Decision Support Systems and Processes

The use of DSS represents a systematic approach to often divisive and intractable issues, such as groundwater availability and its allocation. Defined as interactive computer models, DSS incorporate data relative to a problem and, through programmed analyses, aid the formulation and selection of an appropriate management strategy. The development of a DSS is inherently systemic and multi-disciplinary which differs from traditional analytical approaches that are discipline specific and tend to isolate variables. In addition, the design and development of DSS benefit from engagement and participatory inclusion of stakeholders and decision makers.

Research into the behavior of decision makers demonstrates that the complexity of many decision problems quickly outstrips a decision makers' unaided cognitive capacity (Gregory et al. 2005). Complex socio-technical decisions, such as those needed for groundwater management, are based on large quantities of evidence that is frequently assembled and analyzed by multi-disciplinary teams. The meaning and implications for developing management strategies or actions are evaluated and compared through the eyes of stakeholders. DSS that combine aquifer performance and groundwater governance, as shown in Fig. 25.1, create a more transparent lens through which complex groundwater problems may be viewed without overwhelming stakeholders.

Decisions about aquifer yields are the most common to groundwater problems, though a wide range of other common decision making contexts exist. A non-comprehensive list of examples includes decisions about groundwater availability, such as defining acceptable pumping limits, pump locations, or determining the influence of pumping on threshold flows for groundwater dependent ecosystems (Chap. 15). Another segment of decision contexts include groundwater quality decisions (Chaps. 14 and 15), such as those related to remediation and risk prevention. And decision contexts related to groundwater monitoring stations, sampling locations, or waste management are all good examples of the numerous sets of decision contexts that cross sectors, from industrial to environmental management or domestic and agricultural use cases.

The following sections delve into a more detailed discussion of performance, governance, and decision support elements as they relate to groundwater applications.

## 25.3 Data and Modeled Attributes for Aquifer Performance

Information and knowledge management challenges abound in groundwater sciences. Every DSS is built using datasets and models that represent the problem domain and key elements of interest to decision makers and stakeholders. The domain of hydrogeology is comprised of significant data collections that span spatial and temporal scales across many orders of magnitude with variable resolutions (Narasimhan 2005).

While the scales and extent of groundwater information are vast, the datasets often are sparse considering the complexity in the systems. The resultant uncertainty, paired with inherent variability in groundwater systems limit the predictive value of groundwater models that form the core of decision support systems (Chap. 28). Regardless, offsets of parameters are derived from direct measurements and field observations to quantify and describe groundwater system behavior. These data are used by groundwater modelers to populate, extrapolate, and define a numerical simulation to represent the natural behavior of aquifer systems. Modeled outputs then form the core information for any undertaking in integrated groundwater decision support.

While groundwater modelers are concerned with the low predictive value of numerical simulations for aquifers, from a DSS perspective the focus revolves around (1) linking groundwater with ancillary components in the integrated models (e.g. land use, climatic conditions, and surface water, etc.), and (2) communicating the level of uncertainty as it relates to the decision context.

### 25.3.1 Natural Hydrogeologic Attributes and Uncertainty

Identifying natural attributes of a groundwater system is a vital step in determining a method for calculating relevant performance indicators for decision contexts related to both groundwater response and linkages with ancillary or related aspects for integration.

Parameter uncertainty is a key consideration for assuring that the representative groundwater model reflects actual aquifer behavior. Hydrogeologists have established a myriad of approaches for addressing uncertainty with domain-centric groundwater models (Matott et al. 2009; Banta et al. 2006; Doherty and Skahill 2006; Doherty 2003, 2004; Hamby 1994; Hill 1998; Poeter and Hill 1998). Yet direct assessment and treatment of uncertainty as it relates to integrated groundwater models, such as those that inform DSS applications are less common and recent (Guillaume et al. 2012; Guillaume and Pierce 2011). Integrated modelling is beginning to establish methods and approaches to creating and testing IAMs (e.g. Bennett et al. 2013) and groundwater modelling practice reflects these advances. A key issue is the problem of the low predictive value of groundwater models, particularly when they are combined within an IAM, and the central element of concern is related to the variables and parameters that are used to define the systems of interest, or the attributes. The measurements used to describe and monitor a groundwater system serve as the basic units of knowledge that define performance for decision problems. A natural attribute, defined by Keeney (1992), is a measurable quantity or criterion that has a common interpretation and can indicate the level of achievement of goals or objectives. A review of natural attributes that are common to hydrogeologic problems, compiled by Pierce (2006) and shown in Table 25.1, reveals approximately 37 measures, variables and descriptive parameters.

The units of information shown in Table 25.1 are central to an ontology and scientific understanding of groundwater, as well as being core to the design of groundwater-related decision problems. For example, defining an actual rate of yield or extraction rate, along with primary natural attributes, must begin with the master equation for hydrology, where changes in storage (S) over time (t) can be defined as the difference between inputs (such as recharge) [I(t)] and outputs (such as discharge) [O(t)]. Determining the response of an aquifer to variations in any one of the variables for this equation is key to defining the volumes of groundwater that may be available for extraction. In turn, defining groundwater availability is a quintessential hydrogeology decision problem (Pierce et al. 2013) that may be bounded by limiting constraints for population growth, water demand, and total use of the resource for example.

Natural attributes provide the cornerstone for quantifying and valuing groundwater resources and for developing integrated groundwater management strategies. The natural attributes also serve as the parameters that represent groundwater response in simulation models. Collecting the information needed to understand and model groundwater systems is a necessary first step to decision support.

A DSS links together raw data, empirical calculations, numerical models, and other qualitative factors to analyze decision problems. DSS can help decision-makers conceptualize a problem in a new way, as well as allowing for the rapid conversion of the vast sets of data typically associated with groundwater problems into understandable reports that can provide guidance and insight (Kersten 2000).

## 25.4 Addressing Stakeholder Perspectives for Groundwater Governance

While a great deal of data may exist to inform appropriate analytical or numerical analyses for groundwater resources, the ultimate influences of scientific uncertainty and the issue of complexity require the inclusion of stakeholder perspectives and concerns. Moreover a primary problem as far as DSS is concerned is the communication of this uncertainty to stakeholders and decision makers. The value-based considerations that can only be gleaned from interactions with stakeholders must guide the identification and prioritization of management options that fit with available scientific knowledge and social concerns. In fact, modeling efforts that engage qualitative methods and stakeholder input tend to create more informative problem formulations than traditional efforts without stakeholder advice (Li et al. 2013). These participatory processes are frequently referred to as a co- design and co-creation approach.

Decision support provides a mechanism that interactively bridges the theoretical and methodological gaps between physical systems, analytic outcomes, knowledge interactions and interfaces with users, as well as providing computational support for science-based exploration, dialogue, and/or deliberation. Research on applied, participatory, decision support recognizes that science dialogue is simply another means of communicating ideas or knowledge (Welp et al. 2006) and provides rich qualitative inputs for modeling of complex problems.

Application of fundamental scientific and engineering principles alone can identify a set of management alternatives that are efficient across a number of performance metrics. Yet, technically sound solutions may, in fact, yield options that lead to an unacceptable political price (Allan 1999) because without the aid of a decision support process they neglect social values and process. The Murray-Darling River Basin provides a real world example where farmers protested a technically sound water plan that was unveiled by the Australian Government without adequate stakeholder consultation (Sullivan 2014). Therefore, approaches that recognize the difference between the measurable components of physical systems and the underlying values and preferences that influence management decisions are also needed. Clearly delineating the objective components of a problem from the value-based, or subjectively-judged, components is crucial to assure a final set of decisions that can be implemented without exacerbating disputes (Focazio et al. 2002). For example, a strategic path forward might include efforts to strengthen institutional capacity for managing over-pumped groundwater resources in order to prevent irrevocable damage to an aquifer system. Such governance depends on effective communication with, and advice from, stakeholders and water users.

Effective communication about decision problems follows a recognized set of conventional stages (Mintzberg et al. 1976):

1. Problem formulation or definition
2. Identification of decision objectives
3. Generation and analysis of options
4. Choice of a preferred option
5. Implementation
6. Monitoring and feedback
7. Iteration and problem redefinition.

Groundwater decisions frequently involve a distributed set of stakeholders who need assistance to work through the various stages of decision making and DSS may be of assistance at any of these stages or for multiple stages. Decision support for groups includes processes that enable cooperation among decision makers and stakeholders, while assuring that each participant has a clear stake in the problem that needs to be solved and guides the group towards a shared vision.

Processes may range from informative to strongly normative approaches. Informative approaches attempt to improve the quality of a decision by providing information to help decision makers analyze a situation and assess alternatives. Normative support aims to recommend options based on expected outcomes, rather than strictly explaining information or knowledge.

Regardless of the approach, there is broad agreement that successful processes engage participants and build capacity (van Kerkhoff and Lebel 2006). Consensus building remains the dominant process for creating a shared vision with participatory engagement. Systems thinking (Chap. 24) frequently informs the development of group goals, targets, and criterion. In the context of groundwater governance, consensus yield is a concept that is used for the most common decision making context for groundwater whereby the acceptable range of extraction from an aquifer is bounded by the preferences of affected stakeholders (Pierce et al. 2013; Mace et al. 2001). Consensus yield has become a recognized concept within hydrogeology, yet there are many instances and decision contexts, as discussed previously, where decisions about groundwater and the systems that are naturally linked, or integrated with aquifers, are aided by DSS applications. The preference sets and prioritization of candidate solutions then defines a feasibility space within which technically viable strategies for operational yield and management can be designed. Figure 25.2 shows a conceptualization and example of mapping aquifer performance with the overlay of stakeholder preference points to define a feasible solution space (modified from Pierce 2006). It depicts the intersection between the integrated system response measures, or performance metrics, as generalized groundwater storage response to pumping, and defines the feasibility space.

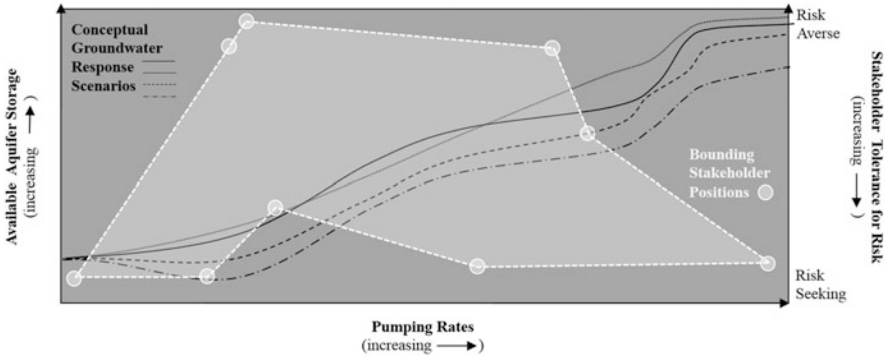


**Fig. 25.2** Conceptual mapping of a feasibility space as defined by hypothetical aquifer performance across multiple scenarios bounded by hypothetical stakeholder preference points (Modified from Pierce 2006)

Framing the problem is a pivotal aspect for capturing principle stakeholder concerns, as well as defining the initial terms of focus for negotiation or deliberation (Chap. 24). Bridging the gap between problem formulation stages and groundwater model development provides an area with potent research potential and opportunities to improve the applicability of research products to real-world groundwater management problems (Borowski and Hare 2007).

## 25.5 Decision Support Systems: Background and Types

As research related to science-based decision making has evolved, increasing levels of insight and understanding are expected to be generated from the application and use of DSS. The field of decision support is constantly advancing at the boundary between theory and application. Theoretically DSS research begins with the premise that improving knowledge management will result in superior outcomes for decisions. For that reason, DSS development activities that target improvements in knowledge management are expected to foster meaningful advances when the DSS are deployed in practice.

Proponents of DSS further claim that activities striving for the most advanced levels will achieve effective knowledge management leading to the generation of 'new' knowledge. The history of DSS development provides a foundation from which to create concrete applications in a specific domain. In assessing DSS case studies that include groundwater, it becomes clear that the level of effort for applying DSS knowledge is significant even while we are able to preview from the broader DSS literature what future advances may achieve.

### 25.5.1 The Emergence of Decision Support

While some practitioners credit Simon (1960) with the presentation of basic management decision processes, Little (1970) was the first to define a DSS as part of the concept of decision calculus. The first international conference on DSS was held in Atlanta, GA in 1981 (Power 2003). DSS literature recognizes that DSS models are simplified representations of problems addressed within a society that assist with the development and evaluation of alternatives. They use multi-objective planning to simultaneously consider various aspects of the decision-making paradigm (Haith and Loucks 1976), such as environmental quality, optimization, and economic cost-benefit analyses.

Since the inception of DSS, theories and applications have evolved to ever more sophisticated approaches over time by leveraging technological advances and transitioning toward improved functionalities and applied competencies on a case-by-case basis.

In the context of groundwater science and governance, the epitomy of groundwater DSS applications will communicate the extent and influence of scientific uncertainty while also enabling interactive deliberation among a plurality of stakeholders. In effect, an idealized DSS for groundwater will provide an advanced level of negotiation and facilitation support. Progressing from fundamental DSS applications to a full DSS with the capability to support live negotiation among groups of stakeholders requires a series of transitions that have been characterized by Kersten and Lai (2008). The progression of DSS types, depicted in Fig. 25.3, identifies transitions among DSS types that range from passive to active, and ultimately proactive applications with the relative level of effort that is necessary

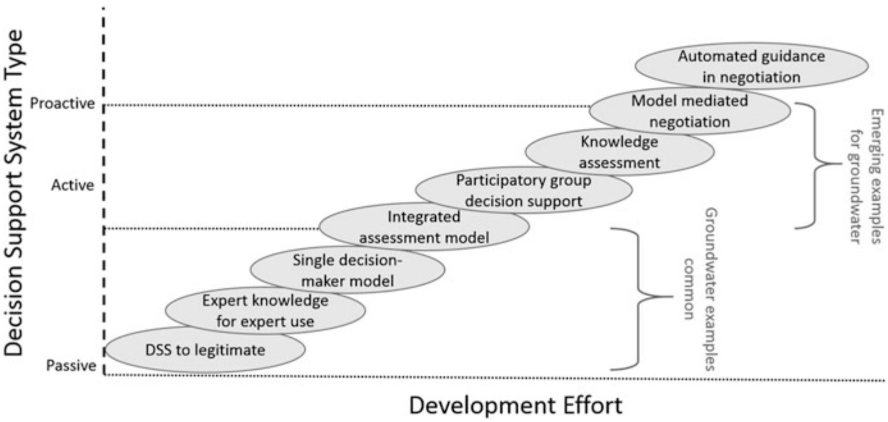


**Fig. 25.3** Evolution of decision support systems for proactive support of science-based deliberation and negotiation (Modified from Kersten and Lai 2008; Pereira and Quintana 2002)

for development, (modified from Kersten and Lai 2008; Pereira and Quintana 2002). The concept of DSS types and tiers (Kersten and Lai 2008) is helpful for assessing the state of use in groundwater cases. The following sections describe the DSS types and evaluate the state of groundwater DSS through this lens.

#### 25.5.1.1 Passive

Passive DSS are tools that aid communication, calculation, and visualization in direct response to the input of a user. These systems augment users' ability to interact or analyze information, but interactivity is limited to direct selection and specification by a user (Kersten and Lai 2008).

The majority of groundwater modeling and management applications reported in the literature can be considered passive type systems from a DSS perspective. Case examples for integrated assessment that include groundwater are beginning to emerge (for example see various cases listed in Table 25.2), with the most advanced case studies transitioning from passive to active style applications.

#### 25.5.1.2 Active

DSS assistance that helps users formulate, evaluate, and solve difficult problems is considered an example of an active system. Active systems provide utilities that support construction and processing of solutions for users (Kersten and Lai 2008). Active DSS may include some automatic knowledge capture or search techniques.

Integrated models and assessment provide the transitional DSS type between passive and active. Jakeman and Letcher (2003) discuss the basic features of integrated assessment models (IAMs), yet little consensus on a generalized framework for the use of IAM's within decision support contexts has been achieved (van Evert et al. 2005; Mysiak et al. 2005). Various approaches and frameworks are presented in the literature (Villa 2007; Khaiter 2005; Moore et al. 2004; Rahman et al. 2004; Sydelko et al. 2001; Argent and Grayson 2003; Segrera et al. 2003; Leavesley et al. 2002). They range from: generalized modeling frameworks that are more accessible to non-programmers but limit specific model implementation; to model-specific frameworks, or implementation-level frameworks, that require a higher level user group, usually with programming experience and result in increased development effort.

#### 25.5.1.3 Proactive

Systems that can evaluate aspects of a decision problem independently with the ability to provide feedback to facilitators/mediators and users during a negotiation process are proactive. These systems are similar to active systems with the addition of facilitator and mediator centric utilities, as well as algorithms with embedded assessment of user inputs in order to derive, or recommend, alternative options. Proactive DSS are expected to provide capabilities to aid group facilitation or mediation, along with the ability to access and use information in real time for the purpose of supporting the facilitators or negotiators. Proactive systems will make suggestions and critiques for improving the outcome of a DSS supported deliberation or negotiation.

**Table 25.2** Decision support or analysis projects with groundwater features

| Source | Problem | Scale | GW Simulation | Optimization & larger DSS | Objective function | Decision variables |
|---|---|---|---|---|---|---|
| Fienen et al. (2013) | Forecasting changes in sea level rise and groundwater | | SEAWAT model aggregated to Bayesian network | Bayesian network to emulate groundwater response/uncertainty | Propagate uncertainty efficiently for use in forecasts for decision makers | Focus on model performance and calibration; decision model not developed |
| Molina et al. (2013a) | Evaluating climate change impacts over time | Regional: Serral-Salinas aquifer, Spain | Used Post-process to evaluate groundwater response; MODFLOW model | Scenarios tested with an Object Oriented Bayesian Network (OOBN) | Comparative analysis across scenarios and time windows; Extensive list of performance measures based on : Agricultural net profits and aquifer storage; Maximizing Total income, employment rates | Intervention actions, such as water rights purchase, land sale, sale of water for irrigation, |
| Hadded et al. (2013) | Water management generally | Local to Regional: Zeuss Koutine aquifer, Tunisia | MODFLOW | WEAP-MODFLOW link | Demand satisfaction, cost and drawdown minimization | Limited by salinity levels and flow capacity |
| Molina et al. (2013b) | | Regional: El Salobra aquifer, Spain | Lumped parameter representation of the aquifer within a linked hydro-economic model | Object Oriented Bayesian Network (OOBN) for stochastic modeling | Assess groundwater quality control with uncertainty; Minimize nitrate concentration and recovery times | Fertilizer quotas Fertilizer prices |
| Le Page et al. (2012) | Water allocation | Regional: Haouz-Mejjate plain, Morroco | MODFLOW | WEAP-MODFLOW link | Evaluate impacts to regions and identify mitigation options | Principally used to validate modelled aquifer response and sensitivities to parameter change |

| | | | | | | |
|---|---|---|---|---|---|---|
| Moura et al. (2011) | Assess groundwater quality control with uncertainty | Local to Regional: farm and aquifer for case studies; Upper Guadiana Basin; Altiplano, Spain | Lumped parameter representation of the aquifer within a linked hydro-economic model | General Algebraic Modeling System (GAMS) and Object Oriented Bayesian Network (OOBN) for stochastic modeling | Maximize gross margin at the farm level as a function of crop prices and yields; the OOBN added response levels in groundwater | Crop surface<br>Irrigation method<br>Soil type |
| Triana et al. (2010) | Evaluate feasibility and performance of water mangemetn strategies | Regional; Lower Arkansas River Basin | Canal seepage and infiltration to groundwater estimated from a MODFLOW/ MT3DMS simulation | Based on River GeoDSS with an Artificial Neural Network (ANN) used to distribute recharge to groundwater | Comparative analysis of estimated performance with a prioritization structure based on performance with<br>Total Storage<br>Water shortages<br>Compliance with legal compact<br>Impacts to water quality | Water strategy choices include:<br>Total water diverted<br>Use of storage<br>Weighting of priorities (shown in Objective column) |
| Van Cauwenbergh et al. (2008) | Ranking alternative water management options with multi-criteria | Local aquifer to regional watershed scale | Mike-SHE<br>Lumped cell structure | Not clearly described; a simplified water transfer model with limited cells | Minimize pumping costs, recharge, and water transport | Not clearly stated, penalty functions are included in the formulation |
| Pierce (2006); Pierce et al. (2006) | Quantifying Sustainable Yield | Local to Regional: Central Texas, Barton Springs aquifer | MODFLOW or an aggregated Systems Dynamics Model of the same system | Link to TABU global search algorithm and systems dynamics model of ancillary systems | Six Objectives defined with stakeholders<br>Max water allocation and location of pumping; two | Pumping (location and rate)<br>drought policy levels for alarm and critical stages |

(continued)

**Table 25.2** (continued)

| Source | Problem | Scale | GW Simulation | Optimization & larger DSS | Objective function | Decision variables |
|---|---|---|---|---|---|---|
| | | | | | formulations for maximizing minimum spring flow; saturated thickness; total storage | Impervious cover and land use |
| Carrera-Hernandez and Gaskin (2006) | Spatially explicit groundwater modeling | Any | MODFLOW | Link to GRASS for geospatial groundwater modeling | Pure simulation capabilities | Not Applicable |
| Letcher (2005) | Water allocation for a watershed basin | Regional to Large: | Network-nodes linked with surface water sites | WaDSS based on ICMS<br>Applied to Namoi & Gwydir River Basins, Australia | Max water allocation | Not clearly stated, but variable options |
| Recio et al. (2005)[a] | Link hydrogeologic model with econometric for agricultural decisions | Regional:<br>Eastern Mancha aquifer, Spain | MODFLOW, possibly 3-D (not clear) steady state | GESMO | Land allocation for crops; Crop yield maximization | Pumping<br>Head levels<br>Electricity costs |
| Mysiak et al. (2005) | Water resource management (general) | Local to regional | Not specified | MULINO | Multi-criteria weighting applications | Varies |
| Lanini et al. (2004) | Participatory Integrated model for basin study | Local to regional:<br>Herault Middle Valley, France | Lumped parameter model of socio-hydrosystem | (no optimization)<br>Matlab/Simulink | No clear description; Stock and flow/ steady state system | Head – drawdown<br>Pumping<br>natural discharge |

| | | | | | | |
|---|---|---|---|---|---|---|
| Quintana et al. (2005)[b] | Groundwater governance | Local to regional: Herault Middle Valley, France | Not clear, but indicates that a groundwater module included | GOUVERNe or TIDDD (Tool to Inform Debates, Dialogues & Deliberations) | Exploratory decision support with stakeholder participants | Not clearly defined |
| Fredrick et al. (2004) | Contaminant susceptibility | Local: single aquifer, NY | 2-D Steady state AEM | (no optimization) Spatial indexing Drastic method | Minimize pollution potential | Water table levels Drastic scores |
| Aziz et al. (2003) | Optimization link for groundwater monitoring plans | Local: contaminant plume various sites | Linear regression for plumes, empirical data, and simplified models | MAROS | Minimize the number of sampling sites and frequency | Monitoring location and time |
| Fatta et al. (2002) | Landfill leachate impact | Local: Ano Liosia landfill, Greece | MODFLOW/MT3D | ECOSIM : Pilot version / local client–server architecture | Linked simulation models, GIS | No decision problem results reported |
| Nalbantis et al. (2002)[c] | Conjunctive use management | Regional: Athens, Greece | MODFLOW Multi-cell and Lumped parameter models | HYDRONOMEAS: Multi-reservoir system management | Stochastic optimization (limited solution algorithm description) | Pumping |
| Oxley et al. (2002) | Land degradation in the Mediterranean | Regional: Argolida, Greece Marina Baixa, Spain | MODFLOW | MODULUS DSS: 9 sub-models for integrated assessment modeling | Solution algorithm and specific objectives not defined: General problem environmental problem scopes | Mentions as possible: Crop choice subsidy change water management and others |
| Naveh and Shamir (2000) | Groundwater level management | Local: Hula Lake, Israel | MODFLOW with GMS | Spreadsheet model | Microsoft Excel solver optimization add-ins | head levels canal flow rates |

(continued)

**Table 25.2** (continued)

| Source | Problem | Scale | GW Simulation | Optimization & larger DSS | Objective function | Decision variables |
|---|---|---|---|---|---|---|
| Demetriou and Punthakey (1999) | Sustainable groundwater management | Regional: Wakool, Murray Darling Basin Australia | MIKE SHE, 3-D flow | MIKE SHE<br>No optimization | Scenario modeling | mainly crop and vegetation related<br>defined with historic data for scenarios |
| Sophocleous and Ma (1998) | Saltwater intrusion (estimate parameters) | Local: Great Bend Prairie aquifer | 3-D density dependent flow/ solute transport (SWIFT II) | Linear regression (forward, backward, stepwise) | Minimize saline intrusion | Hydraulic conductivities<br>pumping rate<br>Distance to – saline interface<br>Layer thickness |
| McKinney et al. (1997) | GIS-based DSS for River Basin Management (prototype level) | Local to regional: Hypothetical | No groundwater component described | GAMs (General Algebraic Modeling system) | Maximize supply; downstream flow; Minimize salt concentrations; power; import sources | Not clearly stated |
| Latinopoulos et al. (1996) | Engineering supply & remediation | Small : Hypothetical | 2-D Method of Characteristics (1 year) | Monte Carlo; Stochastic programming | sum of Total costs + risk | Broken into costs, failure risks, tolerance |
| Andreu et al. (1996) | River basin planning & operational management | Local and Regional: | Eigen value aquifer response flow module – Segura & Tagus basins, Spain | AQUATOOL | Not clearly stated | Not clearly stated |
| Datta and Peralta (1986) | Alternative selection (Surrogate Worth Tradeoff) | Regional: Grand Prairie, AR | 2-D Steady state Flow | Dynamic Multi-objective optimization (Quadratic & Linear) | Min Cost of Water And Max total supply | Pump location & volume<br>Head drawdown<br>Vol. surface water diverted |

Due to the nature of groundwater systems, decision problems for these resources tend to fall into the category of emergent decision contexts (i.e. problems that are ill-defined and lack a common heuristic for identifying solutions). Groundwater management problems will likely require the application of proactive support of science-based deliberation and negotiation DSS. Application of proactive DSS tools for real world groundwater cases are not reported, yet case studies demonstrate transitions from passive toward progressively more active use of DSS tools for groundwater problems. In the future, the field of groundwater decision support systems can be expected to evolve toward increasingly proactive type DSS.

### 25.5.2 Applications of Decision Support to Groundwater Cases

While distributed groundwater modeling approaches have advanced significantly, their incorporation in decision support processes remains limited, and the inclusion of groundwater cases within IAMs or participatory processes is largely absent. The following sections review the use of decision analytic techniques and decision support as reported in the literature for a range of groundwater problems, most frequently discussed in relation to health and environmental quality concerns. Risk assessment techniques have been applied to groundwater problems associated with petroleum spills, waste site leachates, agricultural contaminants, and radioactive materials control (Correll and Dillon 1993).

Control and management of groundwater supply is a primary topic in groundwater research and application, yet few DSS have been developed specifically to address this topic. An evaluation of decision-analysis with hydrogeological applications was put forth by Freeze et al. (Freeze and Massmann 1990; Freeze 1992) for project evaluation. Freeze's paper was timely, preceding the development of a wide-array of DSS for applications to groundwater, particularly contamination and remediation problems (Camara and Cardoso da Silva 1990; Xiang 1993; Lovejoy et al. 1997), but little work can be found applying the same concepts to aquifer yield. A few lumped system approaches without spatial considerations are reported (Naik and Awahthi 2003; National Research Council 1997), or with dimensional approximation (Miles and Chambet 1995), but these efforts lack the credibility of a distributed groundwater model that has been vetted scientifically. To address this issue, advances in linking groundwater with geospatial utilities are streamlining approaches for incorporating spatially detailed models (Carrerra-Hernandez and Gaskin 2006). Spatially-distributed models have been used for permitting and operation decisions while lumped-parameter models are typically used to evaluate socioeconomic relationships.

Sophocleous and Ma (1998) provide one of the earliest groundwater DSS that evaluates the impact of salt water intrusion on aquifer yield. Since 1997 interest in decision support applications has increased (Jamieson 1997). Table 25.2 presents a summary of the literature regarding decision applications and support systems related to groundwater management. Examples include articles that list specific tools or decision analysis applications, as well as integrated models for environmental decisions that include a groundwater component.

Groundwater decision support systems ought to be capable of providing alternative means for approaching water resource management operations through adaptive management for water resources. Table 25.2 also lists decision support and decision analysis projects reported in the literature with groundwater, environmental, optimization, multi-criteria analysis, and other relevant features.

The examples in Table 25.2 demonstrate progressively higher levels of sophistication in the integration of groundwater in DSS applications, yet groundwater DSS have attained primarily passive type DSS and active type cases are emerging. GESMO (Recio et al. 2005) incorporates a steady-state MODFLOW model to evaluate econometric problems for agricultural use on a regional scale. MIKE-SHE (Demetriou and Punthakey 1999) addresses the problem of sustainable groundwater management, but does not incorporate optimization techniques, and instead pure scenario modeling is used. Hydroanemas (Nalbantis et al. 2002) incorporates stochastic programming to address uncertainty and evaluate conjunctive use problems with an embedded MODFLOW model to simulate groundwater response. Gouverne (Quintana et al. 2005) focuses strictly on policy questions to date and incorporates the media-based input from stakeholder participants, but does not clearly describe the groundwater component of the system. WaDSS (Letcher 2005) addresses the problem of water resource distribution on a regional scale linking surface-water and groundwater through a nodal network.

To achieve proactive type guidance tools for DSS, computational advances in areas such as artificial intelligence, optimization algorithms, real-time sensing, informatics, and science visualization will be needed. In the case of groundwater, it is common for subject matter experts to pair models of groundwater response with optimization algorithms. Yet the most advanced algorithmic support remains limited to use by technical experts with particular emphasis on applications for parameterization of numerical models rather than DSS applications.

Development and advances of optimization techniques are integral to the potential for achieving advanced decision support applications. Reviews of optimization applications for groundwater management (Reed et al. 2013; Singh 2012) reveal that the use of traditional optimization and global search techniques have been applied to support decisions related to quantity and quality problems. For example, the groundwater decision support system (GWDSS) presents a hybridized example for water allocation that includes both simulation-optimization and lumped parameter modelling tools (Pierce 2006; Pierce et al. 2006). Artificial Neural Networks (ANN), such as the River GeoDSS (Triana et al. 2010) and Bayesian networks (Molina et al. 2013a, b; Fienen et al. 2013) present an advanced area of research that leverages algorithms to generate potential candidate solutions. The first report of an immersive environment is implemented for a case in the Sichuan Province, China demonstrating a framework that links virtual environments with models (Zhang et al. 2013). As the algorithms and computing capacity have advanced the problems and approaches have also evolved to increasing levels of complexity.

An important indicator of advances and maturity in the field of DSS applications to groundwater problems will be the replication and reuse of DSS methods and software application tools. The application of Bayesian networks (Moura et al. 2011; Molina et al. 2013a, b; Fienen et al. 2013) across multiple cases demonstrates a replicable methodology, and the WEAP-MODFLOW software tool (Le Page et al. 2012; Hadded et al. 2013) is gaining traction across several applications.

Tools and methods are emerging that provide more generalized approaches to DSS for groundwater with some cases shown in Table 25.2 that can be categorized as active type DSS. The pinnacle of applications for model mediated negotiation, or proactive DSS, will require continued advances in computation and algorithm support to identify tradeoffs and candidate solutions among the myriad of complex alternatives.

## 25.6 Factors Related to Adoption of DSS

The complex nature of groundwater resources often overwhelms decision-makers and inhibits the creation of clear management strategies. The possible number of management permutations can be almost innumerable, even for small scale aquifers, which in current accepted practice results in the inefficient evaluation of management alternatives. DSS can provide the computational tools and methodologies to address the complexity of groundwater problems.

Ideally a DSS will consider scientific knowledge, social process, operational constraints, as well as technology system performance. The potential to improve upon current groundwater management and policy practices through the use of science-based DSS is significant. Yet bridging the gaps to advance toward wide-spread adoption and usefulness of groundwater DSS requires explicitly addressing a myriad of factors (see also McIntosh et al. 2011), such as:

- Financial costs – because implementing a DSS system limits groundwater management districts frequently requires software licenses and staff or consultant time.
- Knowledge to implement – use of a DSS system requires the technical capacity to operate and use advanced software products.
- Adaptability of DSS – every decision situation has contextual elements and situation-specific considerations. DSS systems must be easy to adapt to each case before use.
- Multi-disciplinary team – the range of knowledge and expertise necessary to represent a groundwater problem can be very broad and requires expertise across domains.
- Adequate governance structures – without appropriate authority to manage the resources or infrastructure to support a DSS long-term the likelihood of adoption and use drops
- Trust – DSS deployment depends on trust among collaborators.

Groundwater systems frequently cross political boundaries, are exposed to multiple hazards, and affect a broad range of stakeholder groups. Before DSS can be expected to flourish in groundwater use there is the need to: (1) develop new tools that are increasingly transparent to the user groups; (2) improve the integration of tools into daily use by decision makers; and (3) continue collection of input parameter data and improve data measurement. Successful DSS for groundwater management will need to remain flexible and simple enough to explain to various user and decision-making groups while addressing key barriers to adoption.

## 25.7 Conclusions

Groundwater management involves both the facets of an aquifer's behavior as well as the preferences of its users. Users who presume sovereignty over their water rights and withdraw water to meet their individual social-economic needs without considering potentially adverse consequences to others may be following local allocation norms, even as they create the potential for disputes.

In order to address the projected future demands of society for fresh water, groundwater science must provide adequate characterization of the physical system to assure that policy limits for feasible allocation are achievable. Realistic projections of resource demand require incorporating the preferences of the community that depends upon that resource. The interdependency of community drivers and science-based analyses must be recognized and integrated in order to determine the actual availability of a resource under various management schemes.

DSS can provide a set of applications, methodologies, and tools to identify aquifer sensitivity, evaluate inter-relationships among parameters, test alternative management scenarios, and define levels for decision variables that can guide policy making and, ideally, reduce conflict over the resource.

Aquifer decision support is a multi-disciplinary field of study because it relies upon physical models of aquifer behavior, contemporary groundwater data collection systems, rapidly developing simulation and optimization software, as well as qualitative methods to engage and learn from resource users. While the idea of interactive, knowledge-based decision support for groundwater is straightforward, the combination of technical challenges, multi-disciplinary complexity, and scientific uncertainty create significant barriers to implementation. Today, decision support is experiencing a revival in many fields of interest, particularly land use planning and other physical science disciplines. Whether or not the field begins to take form in groundwater sciences will depend in large measure upon the ability of the theoretical techniques to live up to conceptual expectations of the users and the ability of researchers to link theoretical advances to practice.

To meet future water demand scenarios it will be necessary for groundwater aquifers to be managed more effectively and sustainably. Current methods used to determine groundwater allocation and management strategies are neither equitable nor efficient, frequently resulting in the over-abstraction of aquifer systems. Decision support systems (DSS) provide a means for water managers to evaluate complex data sets that include hydrogeologic, economic, legal and environmental elements to calculate available yield for aquifers or estimate levels of risk, resulting in improved policies for groundwater management that may, eventually, help ensure the long-term sustainability of water use by society. Water and humans are inextricably linked. As burgeoning human populations stress existing water resources, civilization needs to manage water. This need highlights the inseparable link between scientific knowledge and human interpretation of the environment. Societies interpret the state of the world around them, and take certain actions upon the physical systems based upon that interpretation. As resource constraints grow and the potential consequences of mismanagement increase, improved methods and DSS for people to convert information into knowledge are vital to ensure long-term resource stability.

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# Integrated Groundwater Data Management

# 26

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**Abstract**

The goal of a data manager is to ensure that data is safely stored, adequately described, discoverable and easily accessible. However, to keep pace with the evolution of groundwater studies in the last decade, the associated data and data management requirements have changed significantly. In particular, there is a growing recognition that management questions cannot be adequately answered by single discipline studies. This has led a push towards the paradigm of integrated modeling, where diverse parts of the hydrological cycle and its human connections are included. This chapter describes groundwater data management practices, and reviews the current state of the art with enterprise groundwater database management systems. It also includes discussion on commonly used data management models, detailing typical data management lifecycles. We discuss the growing use of web services and open standards such as GWML and WaterML2.0 to exchange groundwater information and knowledge, and the need for national data networks. We also discuss cross-jurisdictional interoperability issues, based on our experience sharing groundwater data across the US/Canadian border. Lastly, we present some future trends relating to groundwater data management.

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## 26.1 Introduction

There is a growing recognition that many environmental/hydrological management questions cannot be adequately answered by single discipline studies. This has led a push towards a systems view (Chap. 24), which includes integrating many aspects of the hydrological cycle (Chaps. 1 and 3). The push for integration has significant implications for data management. It requires that data are not only well stored, but also well described, easily discoverable and accessible, and in consistent form for use in the different models in an integrated modeling system. The development of the proto-operational Australian Water Resource Assessments (AWRA) (Van Dijk et al. 2011) system in Australia and a similar system under development by the USGS (Alley et al. 2013) are good examples of this, along with many other studies reported in the literature (Schou et al. 2000; Croke et al. 2006; Krol et al. 2006).

In addition to the focus on integration, new technologies in monitoring and computing, such as advances in computational power and storage, have allowed for an increase in the complexity of studies undertaken. For example, groundwater modeling is increasingly being undertaken at larger scales and groundwater flow is being incorporated into earth system modeling – fully coupled biogeochemical climate models – reflecting the growing awareness of the importance of groundwater systems to society. Therefore, there is a growing need to share data across different jurisdictional and groundwater management areas.

All of these factors mean that groundwater data management, and its support of groundwater modeling, is changing rapidly. It is shifting from discrete standalone data management processes and systems, to connected open and shared data systems that support integrated modeling and decision support (Chap. 25). The chapter is organized as follows: first the concepts of data management are discussed, and then current practices with existing toolsets. This is followed up with case studies and last is some discussion on future directions and trends.

This chapter is not directed at organizations that are responsible for data management; rather it aims to inform the research practitioner who is responsible for an integrated modeling study.

## 26.2 Data Management Lifecycle

### 26.2.1 What Is Data Management?

Data management means different things to different practitioners, and often the varying views reflect the differing roles of the actors in the system. The World Meteorological Organization (WMO) Guide to Hydrological Practices (WMO 2008) provides the following definition:

> We define data management as the set of processes or procedures together with a defined workflow and tools, roles and governance arrangements to ensure secure storage ease of discovery and access as well as ensuring the quality and integrity of the data. These data processes and workflows tend to be formally represented in data management models of which there are many examples. In addition, the implementation of a data management model is with a data management plan.

This definition provides the context for following discussion on groundwater data management.

### 26.2.2 Data Management Models

The task for a data management model is to define the data management workflow and process. It does not necessarily define the governance, nor does it specify how things are to be done. These models are typically defined using graphical representation or formal modeling notation such as Business Process Modeling Notation[1] (BPMN). Here we present two data management models.

The first data management model is presented below in Fig. 26.1, and comes from the WMO Guide to Hydrological Practices (WMO 2008). This model describes a data management scheme where the roles, tools, processes and data products are defined in an abstract manner. This model has been subject to significant input from many practitioners, and is useful as a high-level framework for applications such as integrated groundwater modeling studies. The workflow is described by following the sequence of processes from top to bottom, with the tools used for each of the process connected by dashed lines, and the actors performing particular roles are associated with the tools. In the last column, a range of data inputs and outputs are identified.

The second model is illustrated in Fig. 26.2 using BPMN notation. It is taken from the Data Documentation Initiative (Thomas et al. 2009), which defines a combined cycle including data management processes as well as the associated workflow.

The workflow flows from left to right commencing at the "Start" symbol. Each of the rectangular boxes defines a process and the arrows represent transitions through the workflow from one process to the next.

[1] www.bpmn.org.

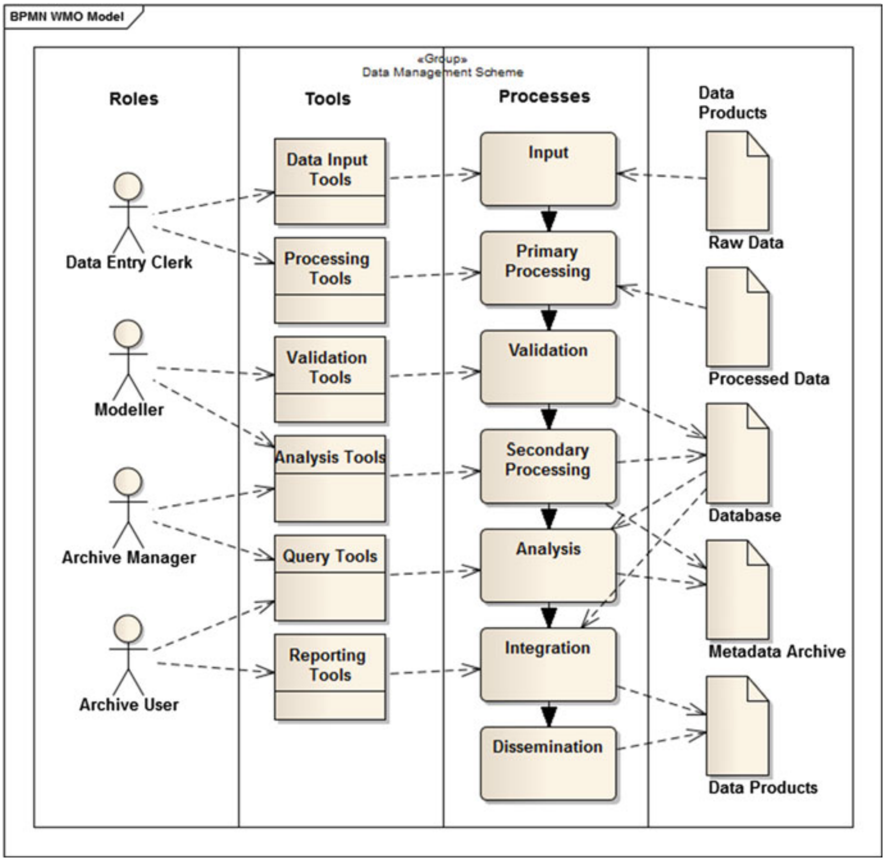


**Fig. 26.1** WMO data management scheme

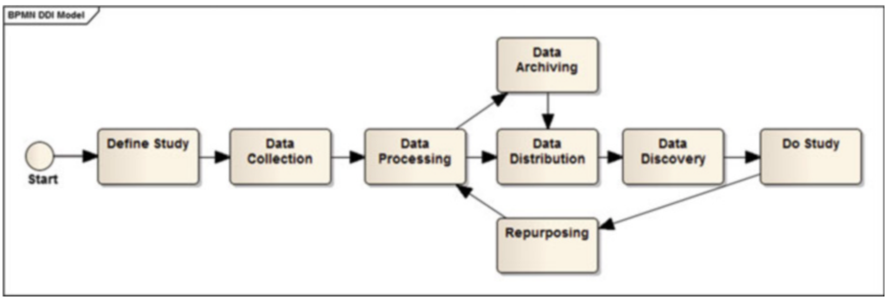


**Fig. 26.2** DDI data lifecycle model

This model can be applied to integrated groundwater studies as follows:

**Define Study** For collection of integrated data, the first goal is to define study objectives, the models to be integrated, and the associated data requirements.

**Data Collection** The next process involves collection of all the data for the integrated study.

**Data Processing** In this step, the data is preprocessed into appropriate resolutions and formats such that it is suitable for the integrated models. Typically at this stage, a number of quality assurance and checks are undertaken.

**Data Archiving** Next, the data is archived in preparation for further distribution and use.

**Data Distribution** Prior to the study being undertaken, the data are made available through a distribution mechanism. This is very consistent with enterprise data management models where centralised data storage is used, either by way of databases or fileservers. These data stores are then accessed for the study by way of a data discovery process. More contemporary methods of data distribution using web services are now gaining favor.

**Data Discovery** In this step, the data are located for the groundwater study.

**Do Study** This is the step in the model where the study is performed. Note groundwater studies, especially modeling studies, almost always are iterative, and this iteration is reflected in the subsequent repurposing of the data.

**Repurposing** The final step in this workflow, takes the data generated by the groundwater study and repurposes it for another use. This could either be another integrated study, or simply another iteration within the current study.

It is worth noting that this data management model can be modified depending upon the purpose of the study and is provided as a general-purpose model. For example an additional feedback loop can be drawn between 'Do Study' and 'Data Collection' if during the study additional data needs have been identified.

### 26.2.3 The Data management Challenge

Data management is successful when data are discoverable, available, accessible, understandable, and usable (Robbins 2012). This perspective comes from the ecological community and their long-term ecological research (LTER) program. It recognizes that successful studies depend on the development of integrated databases and data sets, many of which are collected by different teams over different timescales and are required to be brought together to tackle integrated scientific challenges (Costello 2009), such as integrated groundwater modeling studies. However, while management of data is a core part of the mission of large organizations such as USGS and Bureau of Meteorology in Australia, it is often the case that even within these organizations it is difficult to establish good data management practices in research projects.

Data management is beset with multifaceted problems characterized by social, cultural, and technical dimensions. The social and cultural issues associated with data management are often overlooked and can often be the reason why organizations, research project teams, and individuals, struggle with it.

Leadership heavily influences the culture of an organization, by modeling and defining behavior and values. This is particularly evident in many research projects and integrated modeling studies. It therefore follows that perhaps the most important single driver for good data management within an organization, project or study is the priority placed on it by leadership. This begins with individual practitioners recognizing the value of data, and its management, and cascades to project leaders and senior managers, who include and enforce data management in project plans through policies and adequate resourcing (Costello 2009). Efforts in this area are also augmented by leadership from national agencies such as the US National Science Foundation (NSF) and UK National Environment Research Council (NERC), which now require a data management plan to be prepared with all research funding applications.

### 26.2.4 Data Quality Assurance and Quality Control

The concepts of data Quality Assurance (QA) and Quality Control (QC) are profoundly critical any study. This topic is mentioned here because of its importance, but the reader is referred to WMO 2008 for a detailed treatment of the practical issues and approaches to ensuring QA/QC of hydrological data. In this section we will provide definitions of QA and QC, illustrating the differences, which are not always well understood.

QC is defined as a procedure or set of procedures intended to ensure that data adheres to a defined set of quality criteria, typically accuracy and reliability. These checks are usually done post data acquisition. QA is a more systematic approach to ensuring that the data will meet quality requirements, typically undertaken prior to data acquisition. To illustrate these differences, we will use a manufacturing example. Say a plastic part is manufactured with specific dimensions and tolerance of 10 mm square plus or minus 0.1 mm. A quality control is to check these dimensions with a micrometer to confirm that the part meets specification. In this case the dimension and tolerances are the quality criteria. For data quality control, checks could include bounds checking (not exceeding known maximum or minimum criteria) and that it conforms to some expected distribution and so on.

QA is defined as a procedure or a systematic set of procedures intended to ensure quality controlled data. These are procedures undertaken before data acquisition, intended to improve/ensure quality once checked for. In our manufacturing example, these might include regular maintenance of the machine that manufactures the part, training for the operator, etc. Examples of this for data measurement systems can include instrument calibration procedures, operator training and so on.

QA and QC are usually bundled together as QA/QC without a good understanding of the differences and are commonly now tackled together by organisations implementing a quality management framework such as ISO 9001.[2]

For more information, the reader is directed to WMO (2008, Chap. 9) for details on data processing and quality control.

### 26.2.5 Data Licensing

There is a growing push towards the idea of open data across the research and government sectors, particularly for data supported by publically funded programs. Opendefinition.org provides the following definition: "a piece of data or content is open if anyone is free to use, reuse, and redistribute it – subject only, at most to the requirement to attribute and/or share – alike." Examples of the growing interest in open data are the open data agendas of the United States, Canada, United Kingdom and Australia. These are manifest in data discovery and access portals such as data.gov, data.gov.au, and others. Many of these data initiatives use open data licensing such as Open Data Commons (opendatacommons.org) and Creative Commons (creativecommons.org.au). The intent of all of these open license formats is to maintain copyright with the data creator, ensure attribution, and to transfer risk of use to the user. The interest in Opendata is driven by the assumption that making data freely available generates greater value to society. The authors of this chapter subscribe to this view.

Much data used in integrated studies are subject to a restrictive data license. This is particularly the case in environmental studies where there has been significant cost to collect hydrogeological data, lithological data, and so on. There are potentially other concerns that may limit availability such as commercial interests (eg. storage levels within a hydro-electricity scheme) or potential security concerns. In our work with large scale integrated surface and groundwater modeling, the majority of data have come from state jurisdictions and water management authorities, and is subject to strict licensing conditions. It is often the case for the data to be licensed for a particular study, and in some cases with conditions stipulating deletion once the study is complete (Hartcher and Lemon 2008). Any data management initiative thus needs to be fully cognizant of the many and varied and often strict data licensing requirements.

[2] http://www.iso.org/iso/iso_9000.

### 26.2.6 Data Management and Analysis Tools

Integrated groundwater studies have a specific set of requirements for data types and their specific data management needs. For integrated groundwater modeling studies, these are well described by Refsgaard et al. (2010). Typical data include borehole data containing general descriptions, location, lithology, borehole geophysics, water level and water chemistry. This is supplemented with surface geophysical data, which might include seismic, electromagnetic and electrical data from which the hydrogeology and conceptual models of the groundwater systems can be developed. Most groundwater data management systems have separate tools, processes, and mechanisms for storage of time series, GIS, and spatial data, metadata, and conceptual models.

## 26.3 Time Series Data Management

There exist many commercial time-series data management systems, which specialize in the storage, dissemination and management of surface and groundwater data (e.g. WISKI,[3] Schlumberger[4] and Aquatic Informatics[5]). These types of software packages typically allow ingestion of a variety of data sources including telemetry from automated gages, perform quality assurance, and usually are coupled to integrated analysis tools. They are also able to store a broad set of other hydrological, meteorological and climate data. Most of these systems use relational database technology as the persistence mechanism, which is then attached to a series of tools, as can be seen in the abstract model of a timeseries data management system in Fig. 26.3 below. In this diagram, we map the functional elements described by WMO in Fig. 26.1 above to this abstract model. For these systems, the data output toolsets are increasingly being used to deliver data outside the enterprise using web services and open standards such as WaterML2.0 (Taylor et al. 2013).

This ability to deliver data outside the enterprise becomes very useful for integrated studies and allows time series systems to become part of a web-based data network, which is discussed further below in web-based data management and modeling section.

[3] http://www.kisters.eu/english/html/homepage.html.

[4] http://www.slb.com/services/software.aspx.

[5] http://aquaticinformatics.com.

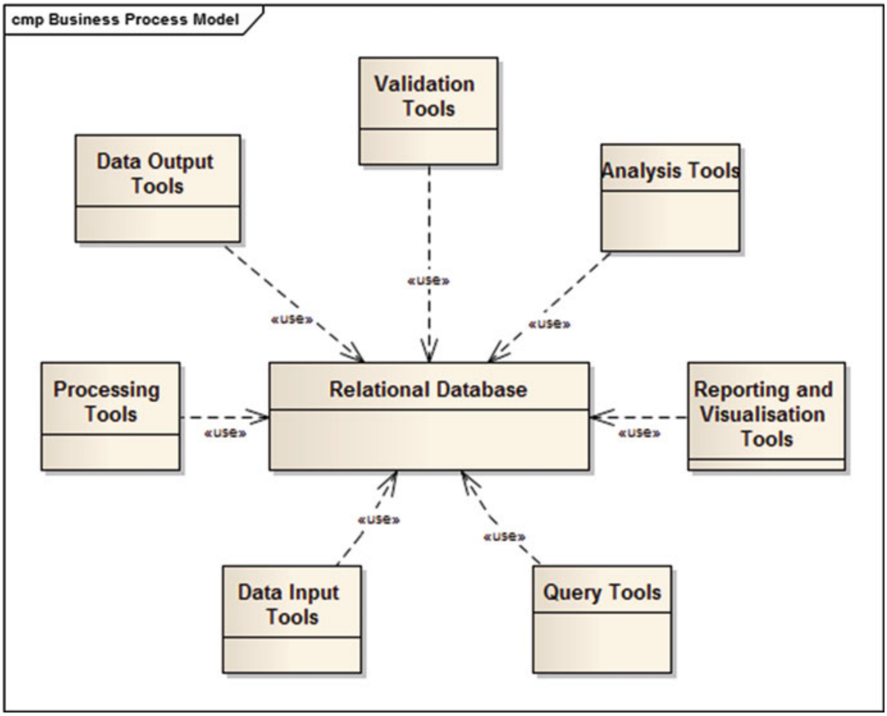


**Fig. 26.3** Abstract model of a time series data management system

## 26.4 GIS toolsets

GIS systems are a core tool for integrated environmental modeling and are widely used (Argent 2003; Gogu et al. 2001; Whiteaker et al. 2006). GIS toolsets are used for spatial and temporal data management, spatial data-processing and analysis, and they can form a software framework for integrated modeling scenarios (Ames et al. 2012).

In Fig. 26.4 above, Argent (2003) describes how GIS systems can be used for integrated modelling application. Two workflows are described, one simply uses GIS for spatial data management (diagram on the right) and the other (on the left) describes a more integrated use of GIS toolsets. In this workflow, the GIS becomes the integration tool, where various modeling applications are created and run. For a good example of this type of workflow, see Gogu et al. (2001).

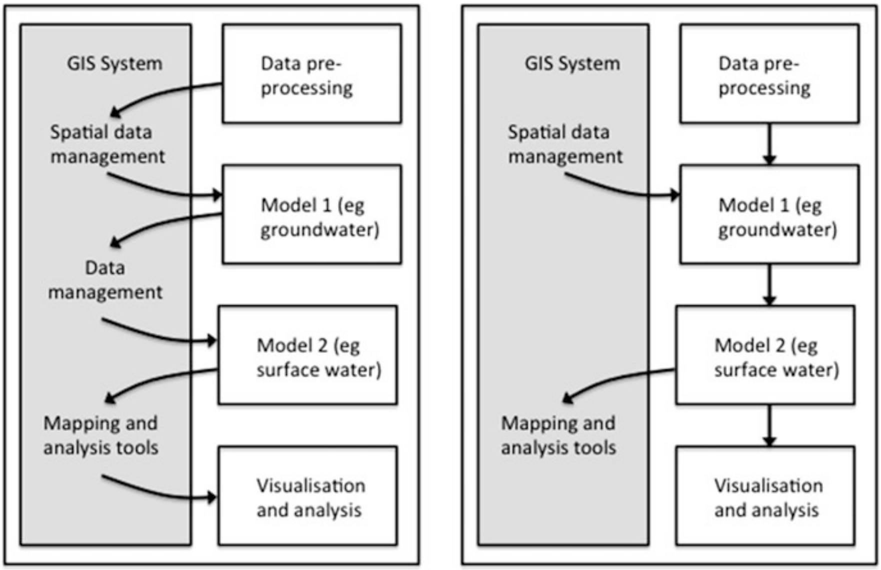


**Fig. 26.4** GIS workflow for integrated modelling after Argent (2003)

## 26.5 Examples of GIS Data models

The widespread use of GIS systems as a data management and data integration tool has led to the development of domain specific geospatial databases, called GeoDatabases. These are optimized for the sorts of data commonly used in geospatial studies, in this case with integrated groundwater studies. These Geodatabase models (Strassberg et al 2004; Jarar Oulidi et al 2009; Chesnaux et al 2011; Yang et al 2010b) represent the features and properties of hydrogeological systems, in ways that allow storage, integration and manipulation of the spatial and time series data. In the hydrology domain, the two most widely used models are ARCHydro (Maidment 2002) for surface water studies, and ARCHydro-GW (Strassberg et al. 2004) for groundwater studies.

ARCHydro is a geographical data model for hydrological systems designed to support a cartographic representation of hydrological features. It is designed to provides a unified model for geospatial and time series data in support of integrated hydrological modeling and analysis (Strassberg et al. 2004). It allows different aspects of the water-resource systems, such as a drainage system, hydro-network and channel system, to be linked to time series flow observations and managed within the GIS system.

ARCHydroGW provides a data model for hydrogeologic units, boreholes and other aspects of groundwater systems that can be used for integrated modelling.

There are many studies which have successfully used these types of models (Whiteaker et al. 2006) in conjunction with GIS toolsets.

One issue that arises concerns unique identifiers in these types of systems (called HydroID in ARCHydro-GW), which identify features in the geospatial databases. Usually these identifiers have local scope, meaning that they are assigned to be unique within a GeoDatabase, and are most usually non-unique when combining or integrating databases. As a result, it becomes difficult to automatically merge databases when conducting integrated studies, requiring significant effort to match or differentiate hydro-geological features based other information.

Another issue concerns the assignment of a fixed geometry to a feature type. For example, a borehole might be represented by a point, in one particular GeoDatabase, and by a line in another GeoDatabase. Thus integrating the different representations between GeoDatabases becomes problematic. This has led to the development of the Hy-Features (Atkinson et al. 2012) conceptual model, in which the features are defined independently of representation. The difference may seem to be esoteric, but defining features in this way allows for easier integration of data for a particular feature type, and greatly eases integrated studies.

## 26.6 Metadata Requirements

For the integrated modeler, the discovery of data suitable for modeling studies always depends on the availability of suitable metadata and an ability to search across it. Most organizations with data management programs will have metadata standards or profiles defined. Examples include the Australian and New Zealand Land Information Council (ANZLIC) in Australia, and the Federal Geographic Data Committee (FGDC) in the US. In general, there is a significant international adoption of the ISO/TC211[6] standards, and many of the emerging national metadata standards are now using ISO as a core, with profiles or extensions as required. Because of this standardization, many tools are appearing which support these standards and leverage them to allow federated searching capabilities. Examples of these include GeoNetwork (http://geonetwork-opensource.org), GI-Cat (http://essi-lab.eu/do/view/GIcat), and Esri Geoportal (http://www.esri.com/software/arcgis/geoportal). In all of these examples, the tools support a number of different metadata profiles and have the ability to harvest metadata records from other catalogs. This federated search ability distributes the responsibility and burden for the generation and management of metadata to data providers, and then allows federated catalogs to be easily assembled and queried by users.

[6] http://www.isotc211.org/.

## 26.7 Conceptual Models

In hydrological modeling the need for a scientific conceptual model is well known (Refsgaard et al. 2010). Though related, scientific conceptual models are distinguished from information conceptual models (discussed in semantics below). Information conceptual models consist of theoretical knowledge (consistent with the scientific conceptual model), such as feature types and scientific theories, whereas scientific conceptual models are essentially re-constructions of a physical area and consist of representations of actual features. Scientific conceptual models provide a description of the agreed understanding of the system under study. Refsgaard et al. (2010) argue for a scientific conceptual model repository to help combine knowledge effectively. We argue that defining both scientific and information conceptual models, and having them discoverable and readily available, is a key requirement for integrated studies.

## 26.8 Web-Based Data Management and Modeling

Integrated studies by their very nature have significant data management and integration challenges. When coupled with the rapidly growing data holdings (for example, in national agencies), an environment is created where discovery access and use of data becomes increasingly difficult. As a result, an interest in interoperability has grown, and practitioners are increasingly looking to the web for help in data management and modeling, such that web-based data access and management is now common place (Granell et al. 2009; Frehner and Brändli 2006). Much of the recent advances in this area have been precipitated by the more than a decade's interest in Spatial Data Infrastructures (SDI; Masser 2010), which has directly led to the development of pan-national standards such as INSPIRE in Europe (http://inspire.jrc.ec.europa.eu), and the construction of associated data networks, including those for hydrology and hydrogeology. In this model of data management, organizations are responsible for management of data and making it discoverable, accessible and available by way of a data network. This approach has significant benefits for integrated studies.

In the next section, we discuss challenges and approaches to building and coupling groundwater data networks, and describe several examples: one example from Canada, two from the US, a unified Canada-US example, and a US example from academia.

## 26.9 Groundwater Data Networks

Groundwater data networks are becoming an important source of data for groundwater studies, due to the increased breadth and depth of their data holdings (Refsgaard et al. 2010). In data networks, autonomous data sources are federated into a composite entity, which behaves as a unified single enterprise. For example, regional groundwater monitoring networks, water well databases, aquifer maps, and other relevant data, are being variously integrated into larger networks in Australia, Canada, and the US (Booth et al. 2011; Brodaric et al. 2011; Dahlhaus et al. 2012). Such networks are typically arranged in some form of distributed architecture, which dynamically retrieves data from original sources, thus ensuring access to current data. They also typically enable users to query and obtain data via a unified common view, shielding users from the heterogeneity of the original sources. In this way, more data, and more data types, are more readily accessed by those studying groundwater, including modelers.

## 26.10 Challenges: Data Interoperability in Groundwater Data Networks

Data access is a key issue faced by all groundwater data users, including modelers, particularly those carrying out integrated studies using multiple data sources. Barriers to data access involve data availability, fragmentation, and heterogeneity: i.e. not all data are available online, and groundwater data are divided unevenly amongst multiple providers, such that the structure and content of the data is quite heterogeneous. This leads to problems in its usage, because data are hard to find, and once found are difficult to exploit due to the immense work required to re-format the data into a common usable structure. Figure 26.5 illustrates an example of heterogeneity in the lithology descriptions of water well databases from two adjacent Canadian provinces: note the differences in language (French/English), structure (one field/many fields), and content (sand/fine and medium sand).

Overcoming the data access barrier thus requires a solution to the alignment of multiple heterogeneous and distributed data sources, i.e. to the data interoperability problem. Spatial Data Infrastructures (SDI) are a leading approach to this problem, and they are actively being adopted by various water data networks, including those for groundwater. Solutions to data interoperability typically require alignment of the data at five levels: systems, syntax, structure, semantics and pragmatics (Brodaric 2007). Ideally, SDI standards are used at each level, and in the water domain these are being developed in coordination with the Open Geospatial Consortium (OGC), the International Organization for Standardization (ISO), and professional bodies such as the World Meteorological Organization (WMO) (Zaslavsky et al. 2011):

| | cle_noseq integer | epaisseur double precis | matprim character vai | fiss_prim character vai | mat_sec character vai | fiss_sec character vai |
|---|---|---|---|---|---|---|
| 1 | 1 | 1.5 | SABL/BLO | INCO | | INCO |
| 2 | 1 | 3.4 | SABL/BLO | INCO | | INCO |
| 3 | 1 | 3.4 | ARGL/GRA | INCO | | INCO |
| 4 | 1 | 2.7 | SABL/GRA | INCO | | INCO |
| 5 | 1 | 0.3 | TERR | INCO | | INCO |

| | materialcolor character vai | material_1 character vai | material_2 character vai | material_3 character vai | topdepth real | bottomdepth real |
|---|---|---|---|---|---|---|
| 1 | | Topsoil | | | 0 | 0.3048 |
| 2 | Black | Muck | | | 0.3048 | 0.9144 |
| 3 | | Medium Sand | | | 0.9144 | 1.524 |
| 4 | | Fine Sand | Silt | Clay | 1.524 | 7.3152 |

**Fig. 26.5** Heterogeneous water well data from the Canadian Groundwater Information Network (www.gw-info.net)

- **The systems level** involves the deployment of standard web interfaces to the data, typically web services such as WFS (Web Feature Service), SOS (Sensor Observation Service), and WMS (Web Map Service), which transmit features (e.g. wells), observations (e.g. groundwater levels), and map images, respectively (Boring et al. 2012; De La Beaujardière 2006; Panagiotis 2005).
- **The syntax level** involves the use of standard data languages, such as GML (Geographical MarkUp Language; Portele 2007), which can be used to encode data.
- **The structure level** includes standard data schema, such as OGC Observations and Measurements (O&M), WaterML2 (WML2), and GroundwaterML (GWML), which are built with GML and constitute a common structure for observations, water time series, and groundwater features, respectively (Boisvert and Brodaric 2012; Cox 2011; Taylor et al. 2013). Standard schemas are typically diagrammed using well-constrained methods, such as UML, and can be expressed in a variety of formats, such as XML.
- **The semantics level** refers to the use of standard concepts and related terms. The terms are typically organized in vocabularies or codelists, and the concepts are typically organized in computational ontologies. Both can be applied to (1) data content, such as common rock type terms and their definitions, and (2) data structure, such as a commonly defined lithology field containing rock type terms. However, they can also refer to scientific knowledge in general, distinct from data, that is to the components of a scientific conceptual model. This includes definitions for the types of entities in the model, and expressions of underlying theories that drive the model.
- **The pragmatics level** includes standard tools and methods, so that data are collected and processed using common scientific protocols.

As an example, the heterogeneous rock type descriptions from Fig. 26.5 can be resolved via transformations of the data at each level: a query in a web browser, for example wells possessing certain rock types, is translated into requests to WFS web services layered over each database (systems); the web services return water well records, by transforming the structure of the databases into standard GWML (syntax, schema), which uses one field to hold rock types, and the content of this field is populated with the rock types in the logs transformed into a standard English vocabulary (semantics). Community agreed protocols are used to determine how rock type terms correlate between the source data and the standard vocabulary (pragmatics). Finally, the results from each web service are integrated, producing a single unified GWML file that is returned to the modeler.

Note that data networks can vary according to where the transformations occur, for example locally at the source, or centrally, and some networks utilize a hybrid strategy that includes local transformations for some network nodes and centralized transformations for the remainder. Likewise, the degree of data centralization can also vary, as evident by the rise of hybrid approaches that use frequently updated central data caches as access points for some, but not all, of the data in a network. Lastly, the location of catalogs can also be centralized, distributed or hybrid; catalogs contain metadata that enable data to be found in the network and that facilitate data transformations, for example by serving local and standard vocabularies and ontologies. However, regardless of the architectural placement of these items within a network, data interoperability cannot be fully achieved without alignment at each of the five levels.

## 26.11 Examples

This section presents five examples. Example 26.1 is the Canadian Groundwater Information Network and Example 26.2 the US National GroundWater Monitoring Network. These are presented as examples of the trend towards large scale national groundwater data networks. Example 26.3 details an emergent North American Groundwater Data Network and discusses how individual networks, if constructed the right way, can be federated into a single federated groundwater data network. Example 26.4 is that of an academic surface water hydrological data network. Lastly, Example 26.5 discusses the use of integrated hydrological data provided from data networks in a national water assessment system. These five examples illustrate approaches that variously utilize hybrid methods for the placement of data, transformations, and related data catalogs.

**Example 26.1: Canadian Groundwater Information Network**

The Canadian Groundwater Information Network (GIN; Brodaric et al. 2011) is a national federation of groundwater data sources managed by Canadian provinces and some federal departments. At present, it contains water well records for most of Canada, monitoring records (groundwater levels) for some selected provinces, and some key regional aquifer and geology maps. As shown in Fig. 26.6, GIN is an example of an architecture in which a centralized approach is used for data transformation and catalogs, and a hybrid approach is used for data placement, that is it is a mix of centralized data caches and distributed data sources such that some data are obtained from the centralized caches and others directly from the distributed data sources.

GIN consists of three tiers. The bottom tier comprises provincial and federal data sources, exposed online ideally via standard web services and data exchange formats, or occasionally via bulk file downloads in non-standard local formats. The top tier consists of potentially many distributed web portals that provide various user interfaces to the data – included among these is the GIN portal itself (www.gw-info.net). The middle tier connects the top and bottom tiers, in that it (1) carries out the necessary transformations between these tiers, and (2) houses the data caches and catalogs required by the transformations. The data caches and

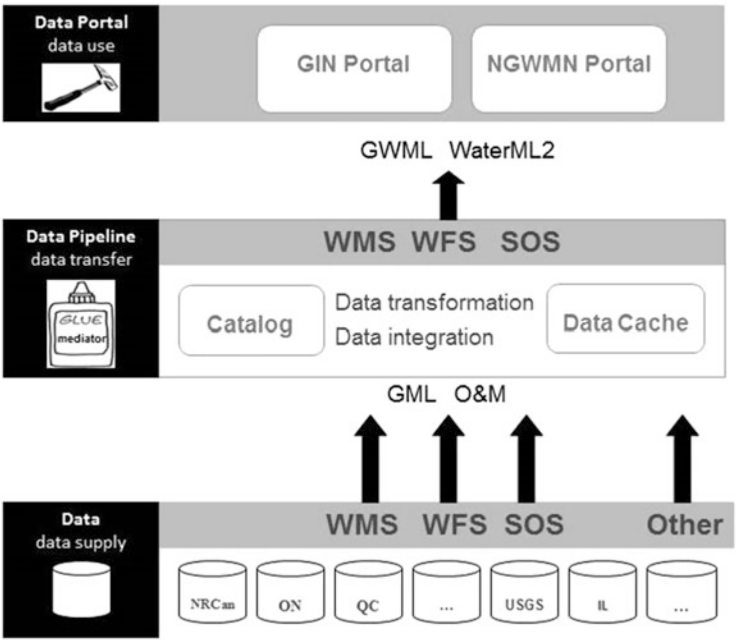


**Fig. 26.6** Architecture for GIN and NGWMN – local data sources in the lowest tier, central data caches, catalogs, and transformations in the middle tier, and distributed web portals in the upper tier

catalogs are updated from local sources either dynamically online via the web services, or manually via file download. The transformations occur in both directions as the middle tier transforms requests from the portals to the local requirements of individual web services or data caches, and conversely transforms the retrieved data to a community standard, either GWML or WaterML2, as required. It also integrates the standardized data, retrieved from potentially multiple sources, into a single unified result, and returns this result to the requester in a choice of several possible file formats such as GML, KML, shape file, ESRI GeoDatabase, or PDF. Significantly, the middle tier is presented online as three web services (WFS, WMS, SOS), which effectively serve as a central data pipeline. Requests for data can thus be made in two ways: through a web portal which issues requests to the data pipeline; or the web portal can by bypassed completely and requests can be sent directly to the data pipeline, for example from an online modeling application.

The GIN architecture has proven to be efficient and effective, returning moderate amounts of data relatively quickly (e.g. hundreds of wells in several seconds), which is adequate for typical usage. Retrieval of large data amounts is enabled via bulk download of pre-packaged files.

**Example 26.2: US National GroundWater Monitoring Network**

The US National Groundwater Monitoring Network (NGWMN; ACWI, 2013) is a recently initiated national federation of US groundwater data. In collaboration with groundwater agencies from US states, the NGWMN links federal and state data in a virtual environment, providing a single online entry point to groundwater data holdings across the nation. NGWMN data include water-well records, water level and water-quality measurements, and references to related aquifers where possible. The NGWMN architecture is very similar to GIN's (Fig. 26.6), utilizing a three-tier portal-pipeline-data architecture, as well as centralized data transformations and catalogs. However, NGWMN differs from GIN in the extent of its data cache, as NGWM caches all data to improve speed of online usage: a data request to NGWMN will thus always retrieve data from its central cache and never directly from the original data sources. The middle tier pipeline implements the same standards as GIN, i.e. GWML, WaterML2, WFS, SOS, and WMS, and also similarly the harvester that populates the cache from local data sources uses these as well as other local standards to ensure that barriers to participation are low. At present NGWMN has completed a pilot stage and adoption continues, incorporating data from more than 20 states and enabling access to this data via an online portal (http://cida.usgs.gov/ngwmn).

**Example 26.3: An Emergent North American Groundwater Data Network**

Coupling of the Canadian and US groundwater data networks is highly desirable, due to the potential for high impact on cross-border groundwater studies. Encouragingly, the coupling of technologies is relatively straightforward, due to the implementation of compatible architectures, and the adherence to common standards across the bottom three interoperability levels (i.e. systems, syntax, and schema), which ensure the use of common web services and related schema. Note that discrepancies at the remaining levels (semantics, pragmatics), which involve differences between vocabularies largely caused by variations in data collection procedures, are managed through data transformations. This is feasible because each network exposes a single data pipeline, which is treated as just another data source by the consuming network. For example, NGWMN is consumed by GIN as if it were another provincial data source, one that requires mapping of vocabularies only, with that mapping taking into account procedural differences.

The coupling of the GIN and NGWMN networks has been tested in two pilot studies carried out in the course of standards development activities at the OGC. In the Groundwater Interoperability Experiment (GWIE; Brodaric and Booth 2010), water level time-series and associated wells across the US-Canada border were found, viewed and downloaded. The Climatology-Hydrology Information Sharing Project (CHISP; Brodaric et al. 2013) was more ambitious, as it involved both surface water and groundwater monitoring gauges, and addressed both water quantity and quality concerns. CHISP enabled cross-border flood risk determination and alerting through dynamic monitoring of gauges upstream from a point of interest, and it also dynamically estimated nutrient loads for any one of the mutually managed Great Lakes.

The GWIE and CHISP studies not only demonstrated that the two groundwater data networks can be successfully coupled, they also directly led to improvements in the networks and to the identification of gaps in the standards, which are subsequently being addressed. Also significantly, they showed that key organizational mandates could be enhanced through the deployment of open standards and the resultant interoperability of the data networks. The end result is the nascent emergence of a North American groundwater data network, which is facilitating access to data for modelers and others in both countries.

**Example 26.4: CUAHSI-HIS and HydroDesktop**

The Consortium of Universities for the Advancement of Hydrological Science (CUAHSI) is a research collaboration of more than 100 US universities and affiliated international research organizations. Apart from its significant scientific contributions, a key achievement of CUAHSI is its hydrological information system (HIS), which enables researchers to publish, manage, and use largely surface water data online (Tarboton et al. 2011). The published data are integrated into the wider HIS data network, which links academic data with major government data sources, such as the USGS, EPA and NOAA. HIS is by far the most de-centralized architecture examined herein, as its data holdings, transformations, catalog and portals are all distributed. Data distribution is achieved, at the moment, using custom "WaterOneFlow" web-services layered over 70 data sources. Data transformation takes place at each data source as an integral component of the web services, and is minimized as standard database structures are encouraged. For data discovery, transformation includes the semantic level, as time series parameters are mapped to a common vocabulary, enabling specific types of data to be identified within the network. However, data retrieval occurs only up to the structure level, as parameters are not mapped to a standard, but served 'as is' from the sources; moreover, data from multiple sources are not integrated into a unified file, but served individually. A central catalog tracks and publishes metadata about the data sources, which can be discovered by online tools. However, in contrast to previous data networks described herein, which are web-centric, HIS emphasizes desktop tools as primary interfaces to the data network. The cornerstone is HydroDesktop, which contains a rich suite of functions for data discovery, management, analysis and modeling. At present, plans are in place to develop HydroShare which will be an online portal that not only incorporates some key HydroDesktop functionality, but will in addition enable many types of collaborative online interactions, most notably the sharing of data and models amongst various research teams (Tarboton 2013).

**Example 26.5: Australian National Water Resource Assessment System[7]**

Following a period of extended drought within Australia the federal government initiated a national plan for water security, enacted as legislation through the Water Act of 2007.[8] An outcome of the Water Act was that the Australian Bureau of Meteorology (BoM) would become the custodian of national water information, and would be required to produce several new water information products, including the annual National Water Accounts and sub-annual National Water Resources Assessments. The AWRA integrated modelling system was developed to support the production of these continental-scale products and integrates three models – landscape processes (AWRA-L), groundwater (AWRA-G) and surface water routing and use (AWRA-R for rivers)

In the proto-operational version of AWRA, where possible, data fetching, pre-processing and loading of input data streams are treated as independent processes, decoupling the modelling system from the data and data management systems. In a complex modelling system such as AWRA, there are many input

[7] Note this section refers to the proto-operational development of AWRA, the final operational version my change in design, scope and implementation.

[8] http://www.environment.gov.au/topics/water/australian-government-water-leadership/water-legislation/key-features-water-act-2007.

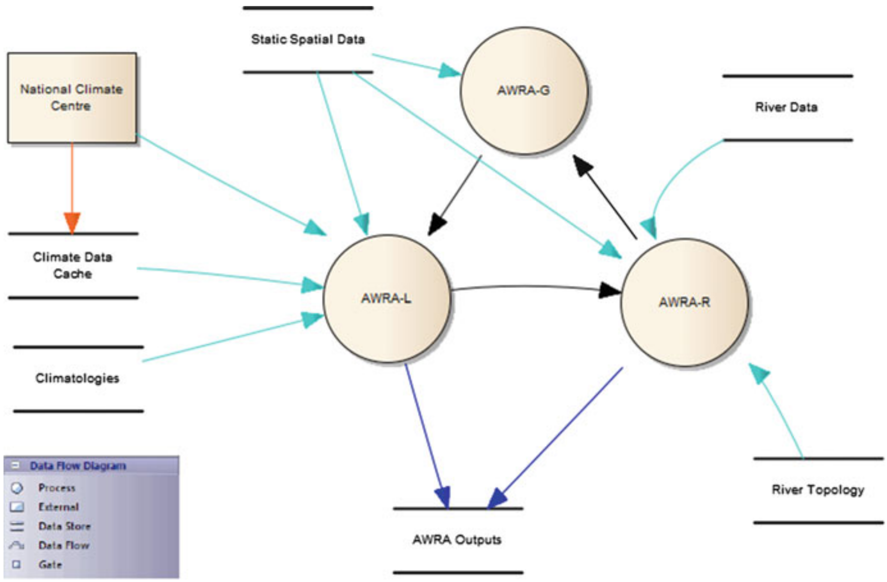


**Fig. 26.7** High level representation of data flows within the AWRA system. Note the barred data sources are internal ad hoc rather than operational data sources. *Orange arrows* are ASCII grids via FTP delivery, *teal arrows* are binary files via direct transfer, *blue arrows* are NetCDF export to THREDDS server and *black arrows* are PI-XML via Delft-FEWS internal data store

data streams, some are standard products and use standardized formats and associated metadata; and they are often supported by a government mandate or service level agreement. These can be considered high trust data streams and have guaranteed availability, and are used in preference to alternatives.

In a real-time modelling system such as AWRA the data fetching is done asynchronously, to both reduce wasted time in the workflow waiting for fetch and pre-processing, and to facilitate future historic runs. The data retrieval process makes use of a local file based data store (Fig. 26.7), which it keeps up to date through both checking for new data, and updating existing data as it is re-published by the data provider following re-processing such as when updated observations become available.

While the fetching of published, operational data streams is preferable from a systems perspective, often the data are incomplete and have gaps either in space or time. In AWRA these gaps are filled through purpose developed data interpolation algorithms or by lookup default values in a post-processing step.

Figure 26.7 shows a high level view of the AWRA modelling system. The diagram shows both the flow of data into and out of the system, and internally between the three major model components. In the original design of the system many of the input data streams were hosted operationally by the Bureau, supported by its new mandate as the custodian of water information. Due to the rapid development of AWRA, and the significant technical and organizational hurdles faced by the Bureau in streamlining the data ingestion process, none of the operational data streams, apart from climate data, are currently available for real-time use by the AWRA system. This has caused complications in the management and updating of the system, and diverted development resources. Once the data network is completed, this problem will be significantly reduced.

Ideally, work on data ingestion would have involved adhering to standards such as WaterML2 (Taylor et al. 2013) for observations, and GML (Portele 2007) for spatial data such as contributing catchments and river network topology. Instead, substantially greater work has been diverted to the collection, checking, re-purposing, re-formatting and management of input data, with all the complications of storage, deployment, duplication, broken provenance chains and a greater number of potential points where errors could be introduced. Once the data services are available through the water data network, AWRA's modular design will allow migration to these new data sources with minimal disruption.

The data sources that will benefit most from availability using a data network approach are those where identity is important such as the naming of river gauges, and those that will need to be extended in their temporal coverage such as river observations. In the current conceptual design of AWRA, the location and identity of river gauges are crucial. The location is used to identify contributing flow from the AWRA-L model and is based largely on the positioning of infrastructure within the river network, rather than by river confluences, although they may be co-located. Over time, as more river reaches are added to the model, gauges are moved or retired; or as the number of gauges used in the model are consolidated, the relationship between river reach models in AWRA-R and the contributing areas used to apportion flow from AWRA-L into those reach models will need to be updated, checked, and incorporated into the model, a time consuming and error prone task. Additionally the mix of points used to define reach models is crucial in the ingestion of observational data such as flow, extractions, diversion and storages, as the identity of those points will be used to resolve the inputs. Currently the network of points, their identities and the related observational data are compiled manually, an even more costly and error prone process than the contributing areas, as the identities are often unique to the agency tasked with monitoring them. The temporal data when collected will often be in different formats that require processing and consolidation, but more crucially the semantic definition of terms is often subtly different, requiring at least a unit conversion, and at worst a conceptual transformation.

Figure 26.8 shows the future idealised data flows into and out of the AWRA system in which the two most important data streams have been replaced by operational web services. These include the network geometry and topology, and associated contributing areas via the GeoSpatial Fabric, and the temporal observation data such as gauged river flow, storage levels and diversion via the AWRIS data warehouse. Crucially, some of greatest headaches in preparing and ingesting input data for the AWRA system will be solved using this approach. The GeoFabric will provide a resolution of identity between the spatial network, the jurisdictional agencies that collect the data, and the AWRIS data warehouse. AWRIS itself will

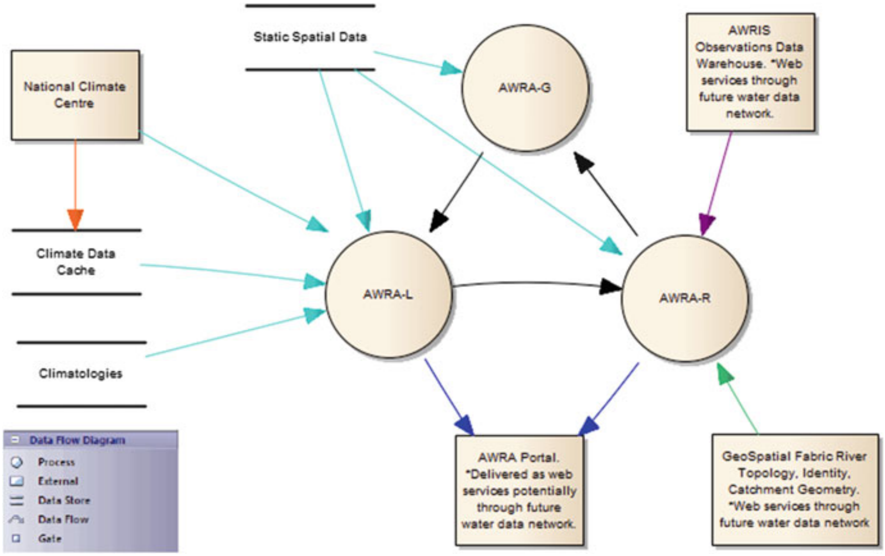


**Fig. 26.8** High-level representation of future idealized data flows for the AWRA system, showing the current ad-hoc data streams replaced by operational services. Note the barred data sources are internal ad hoc, rather than operational, data sources. *Orange arrows* are ASCII grids via FTP delivery, *Teal arrows* are binary files via direct transfer, *blue arrows* are NetCDF export to THREDDS server, *green arrows* are GML via web services, *mauve arrows* are WaterML2 via web services and *black arrows* are PI-XML via Delft-FEWS internal data store

handle the ingestion, consolidation and semantic matching between the diverse sources, as well as proving a trusted data source complete with metadata, and a convenient web services interface supplying data in standardised formats such as WaterML2.

AWRA is a significant national integrated modeling application that has many data management challenges. The current system makes use of many semi-automated steps for the discovery, access, integration and use of data. We have learned that:

- Integrated modeling systems cannot be developed in isolation from the data availability and management needed to support them
- Models need to be managed and governed similarly to data
- Management of data needs to be approached from a dataset by dataset perspective
- A web-based data network would significantly ease the burden of the data management challenge for integrated modeling studies like AWRA.

## 26.12 Discussion of Future Trends

As noted above, it is becoming commonplace to deliver groundwater data online, typically via web services, and to incorporate such data into groundwater studies and modeling activities, also variously occurring online in workflow environments. The totality of these online resources and activities is often referred to as cyber-infrastructure. We anticipate that for integrated modelling studies the cyber-infrastructure paradigm will continue to evolve and grow, likely exponentially.

Furthermore, as cloud-computing technology is also becoming commonplace, it is likely that the processes of data storage, management and integration will occur within the "cloud" (Yang et al. 2010a). This essentially outsources the provision of the hardware side of the data management challenge, with expected gains in efficiencies, reduction of costs and potentially risks. We expect that cloud-computing technology will become an important enabler for delivery of integrated groundwater data in data networks.

Open standards (data and services) are likely to become more common-place with some good current examples being GWML, WaterML2.0 and the underlying GML and XML formats.

Finally, linked data implementations will continue to evolve and grow. Linked data is a term which refers to a set of standards and approaches for publishing and connecting data on the web (Bizer et al. 2009). Linked data is made available on the web in a standard format, usually RDF, which enables links to other datasets, or contextual data including metadata. Because linked data methods use the standard web-based linking approach of Universal Resource Identifiers (URI's), it becomes very easy to discover new data and information on the web. As a result, linked data methods are migrating from the research community and starting to become mainstream, albeit with varying levels of conformance to core linked data principles (Hogan et al. 2012). Examples are appearing in a number of countries, such as the UK location program (http://data.gov.uk/location), in which the identity of features and their corresponding properties can be easily determined.

Two related issues remain a challenge for linked data – these are particularly evident in the water domain. The first is the massive volume of data stored in legacy databases: because linked data approaches, at the moment, almost universally deploy RDF as a format, it still remains a research objective how best to layer linked data methods over non-RDF databases (Marjit et al. 2013). The second associated issue concerns granularity: what is the appropriate granule to be assigned an URI? For example, a particular measurement in a time series, the time series itself, the monitoring site, or even a specific pixel in a remote sensed image? In many of these cases the level of granularity would result in enormous and likely impractical volumes of linked entities. Thus, it becomes important to be able define a certain level of granularity, and have web-friendly mechanisms to delve deeper if required. Nonetheless, we expect that linked data approaches will continue to grow and become an integral part of data networks.

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# Hydroeconomic Models as Decision Support Tools for Conjunctive Management of Surface and Groundwater

# 27

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**Abstract**

Conjunctive use (CU) of surface and groundwater storage and supplies is essential for integrated water management. It is also a key strategy for supporting groundwater-dependent ecosystems, and for adapting water systems to future climate and land use changes. CU has become increasingly sophisticated and integrated with other innovative and traditional water management techniques, such as water transfers, water reuse, demand management, and aquifer remediation. CU adds value for society (increasing average yield and reliability) but can also induce costs to some parties, such as damaging senior water rights of surface water users when pumping from the aquifer reduces streamflow. Groundwater overexploitation also can produce a host of undesirable economic and environmental impacts. Successful CU implementation typically requires changes in infrastructure and operations, but also changes in institutions and institutional arrangements to offset potential third party costs and protect ecosystems. This chapter analyses first the management and economic implications of CU,

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addressing advantages, costs and limitations, as well as the potential contribution of economic instruments to the conjunctive operation of groundwater and surface storage and resources. CU management models are then classified according to the CU problem, their formulation and solution techniques. Different applications of hydroeconomic models are reviewed in a wide range of CU problems. A few applications are discussed more in-depth, using cases from California and Spain. Then, we discuss the relevance of these models in decision-making, and the policy and institutional implications. Finally, we address limitations and challenges, and suggest future directions.

## 27.1 Introduction: Conjunctive Use Overview

Most regions in the world depend on a mixture of surface and groundwater to supply their water demands. This mix of supplies is especially important in semi-arid and arid regions, where seasonal and annual variability in surface water is more pronounced, but humid regions also have seen increased importance of mixed surface and groundwater supplies as populations, environmental concerns, and water demands increase (Downing 1998). Historically, surface and groundwater sources have largely been developed, managed and used independently. However, as water resources in a region become increasingly exploited, population continues to grow, and water transfers become more controversial, the potential benefits of coordinated management of surface and groundwater supplies offer significant incentives for change.

Conjunctive use (CU) of surface and groundwater resources has long been recognized as essential for integrated water management (Buras 1963; Burt 1967; Coe 1990). CU implies the coordinated management and use of surface and groundwater resources, taking advantage of their complementary properties. Although both surface and groundwater storages are used to redistribute water over time to match supply and demands, they differ in storage capacity, recharge and depletion rates, water quality, capital and operating costs, and physical, operational and institutional constraints. Jointly operating all manageable water resources in a region can increase the yield, efficiency, supply reliability and cost-effectiveness for a system. CU is also a key strategy for supporting groundwater-dependent ecosystems (Chap. 13; Kløve et al. 2011 and 2013), as well as for the adaptation of water resource systems to future climate and land use changes (Chaps. 4 and 5; Hanson et al. 2012).

Compared with surface storage, groundwater storage offers vast storage reserves, usually orders of magnitude larger than available surface storage in most watersheds. These reserves can help reduce or eliminate water shortages, acting as a “buffer stock” that provides a reliable, although informal, insurance system (Perez and Gómez 2013) Moreover, the great natural storage capacity of aquifers can be used to store excess surface water in wet periods, increasing ground water levels for use in subsequent dry periods. This could be achieved by artificial recharge techniques (Chaps. 16 and 17), or simply by alternating surface and groundwater use for irrigation, and taking advantage of the recharge coming from river and reservoir losses and the percolation of irrigated water (Sahuquillo and Lluria 2003). Aquifers provide a natural long-term water storage reservoir, without evaporation losses (except from very shallow aquifers). Efficient conjunctive operation increases supply by reducing losses from the freshwater system through reduced flow to the ocean or salt sinks and reduced evaporation from surface reservoirs (Coe 1990).

Groundwater bodies provide additional resources, but also means for water storage, distribution and treatment, which can be combined advantageously with surface water resources and facilities. CU can reduce drainage and salinity problems in irrigated and coastal areas, and water quality improvement is possible with more opportunities for blending water of different qualities and use of soil/aquifer media to treat water (SAT – soil aquifer treatment; see Fox et al. 2001). Initiatives in the US include the use of Aquifer Storage and Recovery (ASR) technologies in Florida, in the ambitious Comprehensive Everglades Restoration Plan (NRC 2012). The costly ASR component in the CU operations was avoided in Al Khamisia et al. (2013) by combining reclaimed water use directly with groundwater to meet irrigation demands. Conjunctive use may also facilitate the integration of reclaimed water to meet urban landscape irrigation demands (green strips and public gardens).

Despite all these advantages, the potential for CU of surface and groundwater has not been fully developed and implemented in many water systems. Traditionally, groundwater has been used only as a backup supply for times of shortage. Perhaps reflecting the bygone eras of their design, most large water supply systems continue to depend exclusively on surface water. Some physical, institutional and legal constraints make implementation of efficient conjunctive use management difficult. Physical and institutional factors promote, shape, and limit conjunctive management (Blomquist et al. 2004).

Conjunctive use operations involve diverse environmental, economic and social aspects, given that alterations in the natural cycle of surface water and groundwater are likely to cause costs and benefits not only to the directs users, but also the neighboring uses. The goals of the CU should be transparent and built with stakeholders' involvement and consensus, to avoid later conflicts. Communication is also critical for success. For example, conjunctive use operations using water banking will affect groundwater pumping costs to both users and neighbors due to the water table fluctuations during the refill and drawdown stages, causing both negative and positive externalities. If not properly taken into account and communicated, these may cause later litigation.

Further opportunities for conjunctive management can be exploited when an elaborate network of water infrastructure, water rights and institutions is present. Examples of these opportunities are found in California, where complex surface and groundwater problems have stimulated development of new approaches for conjunctive use. These approaches are focused mostly on integrating storage and conveyance infrastructure to allow more efficient and flexible water allocation and conservation, to broaden the range of beneficiaries and minimize water conflicts. The contemporary application of CU has become increasingly sophisticated and integrated with other innovative and traditional water management techniques, such as water transfers, water reuse, demand management, and aquifer remediation. The complexity of integrated water resources management in general, and conjunctive use in particular, requires methods and tools for predicting impacts and developing efficient and sustainable strategies.

In this chapter, after reviewing some economic and hydrologic tradeoffs of conjunctive management, we analyze the role of models and systems analysis techniques in the design of efficient planning and management strategies for conjunctive use schemes, using some examples from California (USA) and Spain.

## 27.2 Economic and Hydrologic Tradeoffs of Conjunctive Use

Some general economic advantages of conjunctive use include: greater water conservation, smaller surface water storage and distribution infrastructure, better flood control, ready integration with existing development, less danger from dam failure, and better timing of availability of water for distribution (Maknoon and Burges 1978). Conjunctive use schemes can provide other advantages, such as its adaptability to a progressive increase in water demand at a low cost, and the possibility of temporal overexploitation of aquifers to defer costly construction projects, mitigate the effects of droughts, or alleviate drainage problems (Sahuquillo 1985).

The main economic difference between ground and surface water projects is that, in general, initial investments are much lower for ground water, but operation and maintenance costs are higher. In surface water the initial investment is usually high and the operation and maintenance costs are small. An exception is that surface water treatment for urban uses usually requires higher energy and chemical costs (Sahuquillo 1989). Given the natural water distribution provided by groundwater, its integration in conjunctive use operations improves local supply availability, reducing reliance on external large-scale water transfers. To many regions, including California, this lowers operating costs and risks and increases sustainable operation. The latter also means higher investment locally (e.g. groundwater pumping and recharge infrastructure) contributing to local economic development rather than building large infrastructure elsewhere.

CU adds value for society, but also can induce costs to some parties as, for example, damaging senior water rights of surface water users when pumping from the aquifer reduces streamflow. Groundwater overexploitation can also produce a host of undesirable economic and environmental impacts. Adverse effects of overdraft can include: uneconomic pumping conditions, water quality degradation through induced intrusion of saline or poor quality groundwater, flow reduction in streams, wetlands and springs, land subsidence, interference with pre-existing water uses and water rights and a gradual depletion of groundwater storage (Sophocleous 2003; Zektser et al. 2005). CU is often the best solution to stop groundwater overdraft, transitioning to sustainable groundwater management with the least cost (Harou and Lund 2008). Successful CU implementation typically requires changes in infrastructure and operations, but also changes in institutions and institutional arrangements to offset potential third party costs and protect the ecosystems.

Although often underutilized, economic instruments are often decisive for water management to face increasing water scarcity problems. Water is often underpriced, leading to an imbalance between supply and demand and the unsustainable use of resources (NRC 1997). Water supply and demands vary over time and space, and water prices providing signals during times and locations where scarcity is higher can improve the efficiency of water use (Pulido-Velazquez et al. 2013). Some studies show that it is possible to increase welfare by using pricing to implement a conjunctive management strategy in which price signals encourage surface water use during wet years and groundwater use during dry years (e.g. Schuck and Green 2002; Riegels et al. 2013). In other cases, changes in surface water prices and costs affect the relative value of groundwater, reflecting on pumping patterns, operating costs and groundwater storage. Marques et al. (2006) investigate surface and groundwater economic uses in California, showing that lower groundwater pumping costs relative to surface water resulted in a system failure to internalize groundwater pumping externalities, as users switch to groundwater and aquifer overdraft is intensified. The overdraft raised future groundwater pumping costs, with potentially large economic impacts and risk to the feasibility of conjunctive use operations.

Flexible management of additional conjunctive use facilities and groundwater storage capacity under flexible water allocation can generate substantial economic benefits. CU adds operational flexibility to take better advantage of water market transfers, and transfers provide the allocation flexibility to take better advantage of conjunctive use (Pulido-Velazquez et al. 2004). The added flexibility afforded by conjunctive use reduces stress over the water system, especially surface water reservoirs which can be operated less conservatively when part of the storage is transferred to groundwater. This improves reliability and potential gains to all users, including environmental demands.

## 27.3 Hydroeconomic Models Applied to Conjunctive Use

The complexity of water resource systems requires methods to integrate technical, economic, environmental, legal, and social issues within a framework that develops efficient and sustainable water use strategies. Recent decades have seen widespread use of systems analysis to help on planning and management of water resources. This holistic approach requires identification, analysis and evaluation of the interactions among all components of water resource systems over space and time, considering physical and institutional constraints. Combining economic concepts and performance indicators with the modelling of the hydrologic system and infrastructure (hydroeconomic models, HEM) can provide results and insights more directly relevant for water management decisions and policies (Harou et al. 2009). Meanwhile, the common assumption of “stationary conditions” used in hydrologic and management modelling is nowadays under question in a context of accelerated climate change due to global warming and increasing changes in land uses (Milly et al. 2008), possessing new challenges for the modelers. In this context, hydroeconomic models are better prepared to integrate supply and demand management options to identify promising adaptive portfolios for future conditions.

### 27.3.1 Model Components

Most hydroeconomic models share basic components including hydrologic inflows, water management infrastructure, economic water demands, operating costs, and operating rules (Pulido-Velazquez et al. 2008; Harou et al. 2009). In CU models, we also need to characterize groundwater storage and heads (needed to assess pumping cost variation) and, of course, stream-aquifer interactions where significant for the management model and at the required level of accuracy (see Sect. 3.2). Figure 27.1 conceptually represents an HEM applied to conjunctive use management, illustrating the main components, modeled processes and results.

Water resource systems are often conceptualized as a flow network comprised of nodes, without (e.g. confluences or diversions) or with (reservoirs, aquifers) storage capacity, and links (natural or artificial conduits) with a limited capacity through which water moves in particular directions. A conjunctive use model should integrate surface and groundwater hydrology, as well as stream-aquifer interaction where this is relevant. While in non-economic system models, water demand is usually represented through fixed supply targets that have to be satisfied, HEMs require empirically-estimated marginal supply cost and benefit functions to establish the economic value of water supply to the different in-stream and off-stream

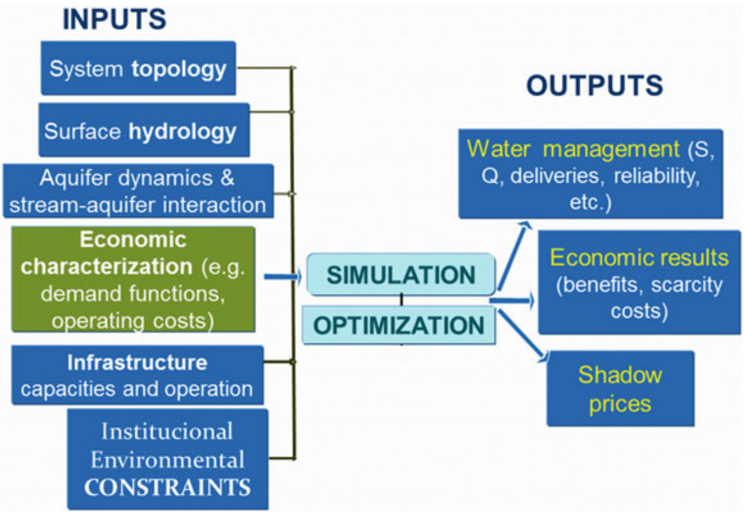


**Fig. 27.1** Conceptual representation of HEM for conjunctive use management

uses. The economic value of water can be characterized exogenously, using different valuation techniques (Young 2005) and external economic models for generating economic demand curves (representing the relation between the quantity of water delivered and its marginal value, ceteris paribus) (e.g. Jenkins et al. 2004; Pulido-Velazquez et al. 2004, 2006). The economic characterization also can be done endogenously, for example including crop production functions in the formulation of the HEM (e.g. Cai et al. 2003). Finally, the model can include different operational, environmental and institutional constraints. These constraints might be relaxed to investigate promising policy or management changes.

A variety of results are provided by HEMs, including flow and storage time series, benefits and costs, marginal economic value of water, and shadow prices for upper and lower bounds. These results can lead to useful conclusions on water allocation and operating decisions, as well as estimates of the economic values of changes in the management and/or the infrastructure capacity, opportunity costs, user's willingness to pay (see WTP coverage in Chap. 21) for water, and other economic and performance indicators (Pulido-Velazquez et al. 2008; Harou et al. 2009).

## 27.3.2 Modelling Techniques

### 27.3.2.1 Hydraulic Management Versus Policy-Allocation Models

CU management models with distributed aquifer simulation are often classified into hydraulic management models, and policy and allocation models (Gorelick 1983).

Hydraulic management models are principally concerned with managing flow, heads and mass transport in the aquifer. For example, optimal groundwater pumping constrained to subsidence control, the control of the evolution of a contaminant plume, or seawater intrusion control in a coastal aquifer. Although these models could be defined with an economic objective (such as maximizing the benefits of groundwater pumping or minimizing pumping cost), often they include other objectives such as maximizing total pumping, subject to the corresponding constraints on the aquifer response. Several examples of these types of optimization problems are provided in Ahlfeld and Mulligan (2000).

In contrast, policy evaluation and allocation models are mainly concerned with the efficient management and allocation of surface and groundwater resources (Bredehoeft 1995). Usually this approach is used in regional agricultural-management problems (Bredehoeft and Young 1983; Lefkoff and Gorelick 1990; McCarl et al. 1999; Pulido-Velazquez et al. 2006) or in large multipurpose regional water supply systems, for example in California (Jenkins et al. 2004; Pulido-Velazquez et al. 2004). The objective function is often defined as maximizing the total economic value of water allocation over time, so that the model will explore the optimal distribution of resources in space and time across the different competing sectors. The economic value of water use is often defined for each use using economic demand curves.

An alternative approach is allocation models in which conjunctive use management is defined based on existing water allocation priorities, without including any explicit economic representation (eg. Fredericks et al. 1998; Pulido-Velazquez et al. 2002).

#### 27.3.2.2 Simulation Versus Optimization Models

Simulation or descriptive models that assess system performance for predefined alternative strategies ("what if" scenarios), permit a more detailed and realistic representation of complex systems, since they are not limited by many of the simplifications needed by the optimization models. In this sense, simulation models are essential for analyzing complex processes of surface and subsurface flow and transport. In groundwater hydrology, the most common models for solving flow and transport equations are based on finite difference or finite element techniques (Anderson and Woesneer 1992). HEM applications usually compare a baseline scenario considering current facilities and operations constrained to current allocation policies with alternative policy scenarios with or without new infrastructure to assess the tradeoffs of a change in system management or design in terms of costs and benefits (Pulido-Velazquez et al. 2008).

Prescriptive optimization models are particularly useful to systematically search for promising planning/management solutions ("what's best" scenarios). A great variety of conjunctive use optimization models are available in the literature, both for hydraulic management and for policy-allocation (as defined in the previous section). Such models typically use linear, non-linear or dynamic techniques with a dynamic balance of relevant quantities (e.g. water flow, contaminant mass), appropriate constraints, and a single (usually economic) or a multiple (e.g. economic, social, target demand) objective (Lall 1995). Network flow programming has been applied for large systems assuming linear or piece-wise linearized responses (Jenkins et al. 2004). Heuristic or nonexact methods like simulated annealing and genetic algorithms have been used for tackling the difficulties of nonlinear nonconvex problems (Rao et al. 2004). Fuzzy approaches allow to deal with uncertainty or account for expert management (Safavi and Alijanian 2011). "Black-box" neural networks approaches have been also employed to simulate groundwater response functions (Karamoutz et al. 2007). There is no general algorithm for solving these problems, but rather the choice of the solver will depend on the characteristics of the system, the scope of the model, the data availability, and the specified objectives and constraints.

#### 27.3.2.3 Representing Groundwater and Stream-Aquifer Interaction in Conjunctive Use Models

Two types of models have been used to quantify stream-aquifer interaction: lumped and distributed-parameter models. Lumped-parameter models use a few parameters to represent the average behavior of the system (e.g., the bathtub model). Most theoretical and empirical economic studies of optimal groundwater management have presented groundwater dynamics using a single-cell bathtub aquifer model, to derive optimal temporal groundwater exploitation (e.g., Burt 1967) or compare optimal management versus competitive myopic solutions (e.g., Gisser and Sanchez 1980; Koundouri 2004). However, other studies have shown that optimal pumping behavior predicted by single-cell bathtub models, which assume that an aquifer responds uniformly and instantly to groundwater pumping, can differ significantly from results of more realistic spatially explicit models with finite hydraulic conductivity (Brozović et al. 2006). Bredehoeft and Young (1970) and Young and Bredehoeft (1972) also showed the importance of an accurate distributed modeling of the aquifer system for conjunctive management purposes, the importance of pumping allocation and the need to manage surface and groundwater as a unit in order to achieve the maximum net benefit. In large-scale River Basin Hydroeconomic (RBHE) models, aquifers are often represented as simple reservoirs with a mass balance equation, often due to the constraints imposed by the applied optimization algorithm (as in network flow optimization models) or the lack of data or more accurate representation.

The linear reservoir model is the simplest model for stream-aquifer connections, and it has been used in simulation and optimization models to indicate promising conjunctive use alternatives at an initial planning stage (e.g., Buras 1963). To analyze a groundwater system with greater accuracy requires a distributed model that explicitly considers the spatial distribution of the aquifer and its hydrodynamic properties, the boundary conditions and the location of external stresses. Analytical solutions have been often useful for a preliminary assessment of stream-aquifer, but most available solutions are developed for ideal homogenous and isotropic aquifers of infinite or semi-infinite extent, idealistic assumptions that can have a significant effect on the accuracy of the results of streamflow depletion (Sophocleous et al. 1995; Pulido-Velazquez et al. 2005). The integration of distributed-parameter models within integrated RBHE optimization models has significant computational implications, and an efficient tool for aquifer simulation is desirable to derive optimal management alternatives or evaluate many alternatives for integrated management over long periods of time. Two major techniques for incorporating distributed groundwater flow simulation within a conjunctive use management optimization model are: the embedding and the response matrix methods (Gorelick 1983; Peralta et al. 1995). A third approach for groundwater flow modeling in basinwide management models is the Eigenvalue Method (Sahuquillo 1983; Andreu and Sahuquillo 1987), in which piezometric heads, flux vectors, and surface and groundwater interactions are obtained by explicit state equations. Unlike the 'embedding method', only the equations that define the control or state variables are loaded into the sets of constraints of the optimization model to simulate groundwater flows, offering computational advantages for the integration of linear distributed-parameter groundwater simulation models within complex conjunctive use models over a long time horizon (Andreu and Sahuquillo 1987; Pulido-Velazquez et al. 2006, 2007a). The Embedded Multireservoir Method also allows quantifying stream-aquifer interaction by simple and operational explicit state equations (Pulido-Velazquez et al. 2005).

#### 27.3.2.4 Ad-hoc Models Versus Decision Support Systems (DSS) Shells

DSSs are interactive computer-based tools to assist in decision-making when addressing complex management problems, integrating simulation and optimization models (Chap. 25). DSSs often involve capabilities of computer assisted graphical design, geographically referenced data bases, and interactive and user-friendly graphical interfaces and tools for input management, results display and analysis. Some examples of DSS with conjunctive use capabilities, such as CALVIN (Jenkins et al. 2004), MODSIM (Fredericks et al. 1998), WEAP (Yates et al. 2005) or AQUATOOL (Andreu et al. 1996), although with significant differences in how water resource systems and conjunctive use are modeled and optimized.

## 27.4 Selected Applications

### 27.4.1 CU Management in Southern California

Southern California's water system imports up to 70 % of its water use, with groundwater being a critical component of the region's water supply. While California's population is expected to increase significantly over the next few decades, on the supply side, traditional imports from the Colorado River and the Owens and Mono Basins are being curtailed, creating a significant water crisis (Chung et al. 2002). The economic-engineering network flow optimization model CALVIN has been used to analyze and compare the economic and reliability benefits from different conjunctive use alternatives (Pulido-Velazquez et al. 2004; Harou and Lund 2008). Results from CALVIN suggest that flexible management of additional conjunctive use facilities and groundwater storage capacity under flexible water allocation can generate substantial economic benefits to the region. Conjunctive use adds operational flexibility needed to take full advantage of water transfers, and transfers provide the allocation flexibility needed to take better advantage of conjunctive use. The value of projected conjunctive use facilities and groundwater storage along the Colorado River Aqueduct, Coachella Valley, and north of the Tehachapi mountains under economically optimized operation of the system is examined. The results reveal reduction of the demand for increased imports into Southern California, suggest changes in the system operations, and indicate significant economic benefits from expanding some conveyance and storage facilities.

### 27.4.2 CU Operations and Irrigated Agriculture Decisions in California

Simulation and optimization models often have been used to support effective conjunctive programs and operations, including approaches with physical stream/aquifer interaction (Gorelick 1983; Peralta et al. 1995; Fredericks et al. 1998; Belaineh et al. 1999) and operating decisions to minimize surface reservoir spills (Schoups et al. 2006a, b). While these approaches help the understanding of surface and groundwater interaction, and how to manage it, its application to local management still lacks representation of detailed users' decisions behind water demands, including water and irrigation technology use under uncertain (stochastic) surface water supplies.

In California, federal, tribal, state and local agencies are responsible for managing surface and groundwater, including water rights regulation, groundwater quality and groundwater management plans. According to the California Department of Water Resources (DWR 2009) water users have few restrictions on groundwater use (except in adjudicated basins) as long as the water is applied to beneficial use. This may cause environmental and economic problems if there is not a proper integrated management of surface and groundwater supplies.

In this context, the hydroeconomic model presented in Marques et al. (2010) addresses farm decisions of water use and crop production. Surface and groundwater are conjunctively managed through artificial recharge to store surface water in the aquifer, and groundwater pumping to retrieve it. Artificial recharge occurs through spreading areas for infiltration, which requires some land dedicated to it. The hydroeconomic model includes surface and groundwater supply and storage, each with its costs, availability, uncertainty and use constraints, integrated with the economic product function of the user, which has water as one of the inputs. This allows the model to capture user decisions on which supply source to use, how efficiently to use it (irrigation technology), when and how much to use.

The approach is based on a two-stage stochastic programming model combining a quadratic crop profit function with permanent and temporary irrigation water use decisions to identify the potential economic gains of conjunctive use operations, and how such operations can be organized. Permanent crop decisions are modeled in the first stage, and annual (temporary) crops are modeled in the second stage, represented by a group of possible hydrologic scenarios (dry and wet years), each with a different water availability and probability of occurrence. In any hydrologic scenario, water can be withdrawn from surface supplies and pumped from the aquifer to irrigate crops, or artificially recharged for posterior use. Conjunctive use operations are represented by additional decision variables for artificial recharge area, volumes recharged and pumped. An intertemporal mass balance equation ensures aquifers recharge matches pumping in the long run to avoid aquifer overdraft. The model maximizes the net expected economic benefit of irrigated crop production in both stages, with conjunctive use operations allowing water to be transferred between different hydrologic scenarios through artificial groundwater pumping and recharge (Fig. 27.2) which is integrated with surface water use and availability.

The model application in Marques et al. (2010) indicated that groundwater availability, price, and conjunctive use operations affect crop and irrigation technology decisions. Groundwater provides a stabilizing effect, increasing permanent (more valuable) crop acreage and expanding annual crops in dry years. Crops with high consumptive demand were not supplied with costly water through low efficiency irrigation technology, and as groundwater supply was curtailed in wet years, surface water was allocated to permanent (more valuable) crops, reducing the

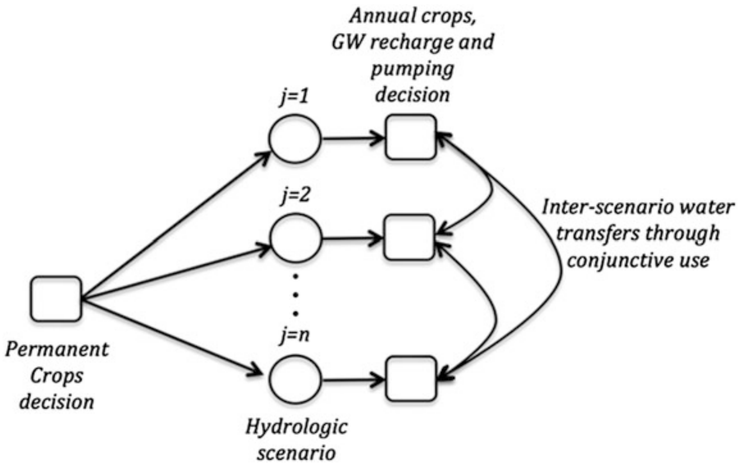


**Fig. 27.2** Diagram of hydroeconomic model decision structure

acreages of annual crops. Artificial recharge was concentrated in very wet years, to take most advantage of the investment in infrastructure. With conjunctive use operations, the gains in income reliability were greater than the gains in the expected net benefit, with a trade-off between both. This information can be useful to evaluate the user's willingness-to-pay for insurance based on risk aversion. While users are likely to increase investment in groundwater pumping capacity, sacrificing some of the total net return gains to build the CU infrastructure, the model allowed the identification of a maximum groundwater pumping capacity investment beyond which no further benefits were expected in reliability or net benefit.

### 27.4.3 Economically Optimal CU in the Adra-Campo de Dalias System (Spain)

In the coastal plain of Campo de Dalias (330 km$^2$) in Almeria province, southeastern Spain, the climate conditions and the application of high-tech agricultural techniques have led to high value crop production, mostly vegetables produced under greenhouses, with a spectacular increase in cultivated land and population, becoming the main factor of economic growth in the province. In this water scarce arid region, the water for the irrigation of the more than 20,000 ha. of cultivated land is obtained from groundwater pumping from the Campo de Dalias aquifer system. The intense use of groundwater for years has led to a significant decline of the water table, causing problems of water availability and quality (e.g. seawater intrusion problems). To reduce overexploitation of the Campo aquifers, water is being imported, beginning in 1987, from the Beninar Reservoir, located in the contiguous Adra River basin.

Pulido-Velazquez et al. (2002) examined different conjunctive use management alternatives for the system in a detailed simulation study. An integrated hydrologic-economic modeling framework for optimizing conjunctive use of surface and groundwater has also been developed for the Adra-Campo de Dalias system (Pulido-Velazquez et al. 2006, 2008). Integrated river basin modeling with distributed groundwater simulation and dynamic stream-aquifer interaction allows a more realistic representation of conjunctive use and the associated economic results. Transient distributed-groundwater flow is simulated by embedding the explicit equations derived from the eigenvalue method (Sahuquillo 1983) as constraints within the nonlinear economic-engineering optimization model. This method provides an efficient approach for aquifer modeling in conjunctive use models, using explicit state equations to characterize the selected state variables. The use of an economic objective function, maximizing the net economic value of water use, provides solutions that optimize economic efficiency in water resources management, while the model constraints guarantee the feasibility and sustainability of suggested operations.

The model results include time series of monthly flow and storage, marginal economic value of water at each location and time step, and shadow prices for upper or lower bounds in reservoirs, stream reaches, canals, and pipelines. These results lead to conclusions on water allocation and operating decisions, as well as estimates of the economic value of changes in the management and/or the capacity of the infrastructure, users' willingness-to-pay for water, and other economic and performance indicators. A systematic approach is provided to estimate time-varying resource and environmental constraint opportunity costs to users at different locations within the system, providing useful indicators for the economic analysis required by the EU Water Directive Framework (Pulido-Velazquez et al. 2006, 2008; Heinz et al. 2007).

## 27.5 Challenges, Benefits and Future Directions

Many choices face the builder of a hydroeconomic conjunctive use model, including the scale and model type for each of the three subsystems (surface water, groundwater and economic demands). Most modeling efforts showcase unique combinations of these because of unique characteristics of the modeled system and the modeler's skills and perspectives. In practice existing models, especially if calibrated and accepted by stakeholders, often influence decisions about how to build an integrated model. A major choice is whether the groundwater model will be lumped (a frequent choice for policy models) or spatially distributed (often appropriate for hydraulic management models). Other decisions, like which water use benefits to include and how to represent their economic values, are shared with all hydroeconomic models and are not particular to conjunctive use hydroeconomic models. This is the case for choosing an appropriate temporal discretization (time-step) and when optimization is used, whether the optimization model should be solved all at once (water users in the model have perfect knowledge of future hydrological flows) or time-step by time-step.

There are several technical challenges to hydroeconomic modeling of conjunctive use of surface and groundwater. For example, in conjunctive use systems nonlinearities may arise due to the physical representation of the system (e.g. nonlinearities due to stream disconnection in stream-aquifer interaction or unconfined aquifers) or the cost structure for surface and groundwater use (e.g. nonlinearity of pumping costs, function of the product of pumping heads and pumping rates at the production well). In simulations models, this can be easily addressed. But for hydroeconomic models using optimization approaches, the potential non-linearities of stream-aquifer interactions, unconfined aquifers or pumping cost functions, introduces difficulties in solving the model and in the verification that the solution is globally optimal. Several researchers have overcome these difficulties in particular modeling efforts (Reichard 1987; Pulido-Velazquez et al. 2006, 2007b, etc.). Still it is a barrier in practice as these are specialty methods known to few practitioners. Hydroeconomic modeling conjunctive studies that use optimization algorithms to solve all governing equations including large sets of discretized spatially explicit groundwater equations may fall prey to numerical difficulties (e.g. Tung and Koltermann 1985; Harou and Lund 2008). Other studies have not reported difficulties in this task but it remains a potential challenge or barrier, particularly for large groundwater models.

Despite early and on-going successes, advanced modeling of conjunctive use in water supply planning and management industry practice is the exception rather than the rule. Often excellent surface water system modeling and groundwater modeling systems exist, but their combined use by industry is still rare globally, with more use in some areas (e.g. California, Spain, Australia, etc.). In water supply planning by utilities, where conjunctive use modeling would be particularly valuable, groundwater is often still represented as an aggregated available supply (e.g. yield). The groundwater field has repeatedly warned against 'safe yield' concepts applied to groundwater (Alley and Leake 2004), yet because adopting this approach means integrating groundwater sources into basic supply–demand models is feasible (e.g. Padula et al. 2013), it persists. Also, many decision support systems built for utility scale water supply planning start with the surface water network and its storage reservoirs; this encourages inclusion of groundwater as another storage node. WEAP (Yates et al. 2005) and AQUATOOL (Andreu et al. 1996) are notable exceptions as they allow linking discretized groundwater models to a surface water resource management model.

The papers and modeling efforts reviewed here show the potential benefits to water management studies of considering hydroeconomic aspects of conjunctive use management systems. These include, amongst others, suggesting how groundwater and surface sources can most productively be used together, how use of each resource economically affects those exploiting the other, how the two resources can efficiently interact within water markets, and how new schemes can have unexpected but significant impacts on other water supplies, either downstream or in the future. These are major benefits and, given the large capital cost of water supply investments, they are in many situations worth the investment. Below we review what future directions this field could take to achieve further scientific and practical impact.

Future potential scientific directions of inquiry are many, starting with the continued improvement of current models and methods for integrated modeling of surface and groundwater and their link to managed systems. Linked groundwater models and surface water simulation can now be linked to single or multi-objective global search algorithms (Reed et al. 2013; Matrosov et al. 2015); this new way to seek efficient solutions opens up many possibilities, including simultaneously considering non-economic objectives. Recent efforts (Yang et al. 2009; Giuliani and Castelletti 2013; Erfani et al. 2013) to move beyond deterministic optimization to represent more realistic behavioral modeling of water users are relevant here. Many optimization modeling efforts reviewed in the paper apply to situations where water markets are relevant; where this is not the case, different computational technologies may be appropriate. Including specific policy investigations (e.g. pricing; Riegels et al. 2013) in addition to water allocation assessment will increase as the tools under discussion are used to assess particular policy investigations.

Several factors could influence growth of hydroeconomic conjunctive use modeling from pockets of excellence (e.g. Western USA, Spain and other localized contexts) to increased global use. The demand from stakeholders and water planners and the availability of easy-to-use decision support systems that model both surface and groundwater systems will likely determine how influential conjunctive use models will be in the future. If their use continues to grow, it is likely that such models with an added hydroeconomic focus will move from academia, their current most frequent institutional home, further into practice.



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# Methods for Exploring Uncertainty in Groundwater Management Predictions

# 28

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**Abstract**

Models of groundwater systems help to integrate knowledge about the natural and human system covering different spatial and temporal scales, often from multiple disciplines, in order to address a range of issues of concern to various stakeholders. A model is simply a tool to express what we think we know. Uncertainty, due to lack of knowledge or natural variability, means that there are always alternative models that may need to be considered. This chapter provides an overview of uncertainty in models and in the definition of a problem to model, highlights approaches to communicating and using predictions of uncertain outcomes and summarises commonly used methods to explore uncertainty in groundwater management predictions. It is intended to raise awareness of how alternative models and hence uncertainty can be explored in order to facilitate the integration of these techniques with groundwater management.

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## 28.1 Introduction

> "Elementary," said he. "It is one of those instances where the reasoner can produce an effect which seems remarkable to his neighbour, because *the latter has missed the one little point which is the basis of the deduction*." – Sherlock Holmes in "The Crooked Man", *The Memoirs of Sherlock Holmes* (1893)

> "How often have I said to you that *when you have eliminated the impossible, whatever remains, however improbable, must be the truth*?" – Sherlock Holmes, *The Sign of the Four*, ch. 6 (1890)

> "You know my methods, Watson. *There was not one of them which I did not apply to the inquiry*. And it ended by my discovering traces, but very different ones from those which I had expected." – Sherlock Holmes in "The Crooked Man", *The Memoirs of Sherlock Holmes* (1893, Doubleday p. 416)'

The issue of exploring uncertainty in model-based prediction can be described through three quotes by Sherlock Holmes. Firstly, any particular model may fail to capture a crucial characteristic of a problem. Hence, prediction needs to involve exploration of 'alternative' models in the hope that one may include the one little point which is important for obtaining a sufficiently accurate prediction. Secondly, because the scientific method cannot prove correctness, prediction of uncertain outcomes needs to focus on eliminating the impossible and incorrect. Thirdly, it is often necessary to use multiple methods – because groundwater management involves a hidden and poorly characterized subsurface, there is no definite way of determining which of many methods will provide the necessary information. Although the world of all possible methods are only within reach of experts like Sherlock Holmes, anybody who deals with prediction of uncertain outcomes will benefit from becoming aware of the approaches available and the principles underlying them. This is the purpose of this chapter.

Addressing uncertainty is an indispensable part of prediction. Groundwater management faces uncertainty on many fronts, in understanding the behaviour of the groundwater system, anticipating possible future climatic, economic or geopolitical conditions, prioritising objectives, all combining to add ambiguity in the

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evaluation of management options. Focussing on the first, it is apparent that scientific research has achieved relative success in reducing this uncertainty, culminating in the ability to approximate the behaviour of a groundwater system using a ‘model’. There are, however, limits to the ability of science. Far from being all-knowing, there will always be recognised and unrecognised unknowns that mean that a model will always be a simplification of reality, and the predictions it makes will always be uncertain (Hunt and Welter 2010; Guillaume et al. 2012).

A model is simply a tool to encapsulate and transparently express what we think we know and illuminate what we do not (Doherty 2011). It is only as good as the conceptualization that was put in it, and when misused can easily result in ‘garbage-in’ producing ‘garbage-out’. In interpreting results, the end-user should only weight their assessment of a model’s prediction by the confidence they have in the model itself. Given the open system being modelled, an end-user and modeller likely will have sufficient confidence in a family of possible models, each of which makes a different prediction (Barnett et al. 2012). Creating an ensemble of predictions of an uncertain outcome ultimately amounts to using many models and their associated confidence to produce a probability distribution, a confidence interval, or simply a set of scenarios which are believed to have utility for the modelling purpose.

This chapter provides an overview of commonly used methods to explore uncertainty in groundwater management predictions. Their common element is that they help produce ‘alternative’ models in which the end-user may have sufficient confidence, even though there may be other models which appear better. The presentation is aimed at end-users of groundwater management predictions, including managers and water users, to help them become aware of the methods available to generate alternative models and hence relate to prediction uncertainty. It may therefore also be of use to modellers to help explain how a technique helps address uncertainty. It is generally accepted amongst modellers that stakeholders ultimately decide whether the accuracy of a prediction is acceptable (Refsgaard and Henriksen 2004). Evaluating prediction accuracy requires understanding how it was produced. In a modelling context, this means being able to critique the reasons why a given model(s) was selected. We encourage all end-users, particularly groundwater managers, to be aware of the different reasoning underlying these methods. We do not expect the reader of this chapter to become an expert. We focus primarily on uncertainty in groundwater models, as a fundamental tool for expressing uncertainty in groundwater management. However, many of the methods can be used with other types of models. We expect the reader may gain an understanding of how modellers can ‘dance’ with a model to explore alternatives. They may be better prepared to participate in judging the value of the information that was put in the modelling, and hence enhance their confidence in the predictions of uncertain outcomes produced. We hope this will consequently help dispel the magical aura and unassailable authority that model predictions often seem to carry, while giving a language for relating the uncertainty that surrounds all predictions.

In order to describe methods to explore uncertainty in a groundwater model, the chapter initially sets the scene by discussing the construction of a clear modelling problem definition, and options for using and communicating predictions of uncertain outcomes. The methods covered include creating alternative models with different input values and different structures (in terms of both conceptualisation and implementation). Other methods select different parameters using statistical properties of data as well as fitting observations of (multiple) predicted variables, or by aiming to test a hypothesis, or estimate the importance of variables. We finish with methods to anticipate surprise by supporting adaptation and exploring the 'known unknowns' of Hunt and Welter (2010). For each method, the general principles tend to be broadly applicable to other types of models, but are illustrated with cases specific to groundwater modelling.

## 28.2 Starting from a Clear Problem Definition

The methods described later in the chapter assume that the scope of the problem has been defined. In particular, this means that there are clear predictions to make (Barnett et al. 2012). Models are a simplification of reality, and therefore do not represent all aspects of reality, but modelling needs to adequately capture the salient behaviour of the system of interest for a given purpose (Jakeman et al. 2006). Knowing how predictions will be used should directly inform the modelling approach because it forms the basis for deciding which simplifications and simulation processes are required in the model, and which can be omitted. For example, prediction of groundwater head is a fundamentally different (and easier) problem than prediction of groundwater transport.

Once a clear set of predictions is identified, the problem is expressed in modelling terms. A model, by definition, is a simplification of a system. As shown in Fig. 28.1, it produces outputs from given inputs, such as rainfall and pumping. The response of the model to these inputs can be modified by changing the value of so-called parameters, such as properties of the modelled aquifer. We need to know what inputs are needed and what outputs are expected. An alternative view of a model is the XLRM framework (Lempert et al. 2003). An end-user may be interested in investigating the effect of different exogenous uncertainties (X) and policy levers (L), and will be expecting that the relationship (R) captured by a model and its parameters will produce certain measures (M) to evaluate them by.

Figure 28.2 shows a more detailed example of the use of a groundwater model as part of a broader integrated model aimed at assessing the impacts of changes in climate conditions and water allocation policies on surface and groundwater-

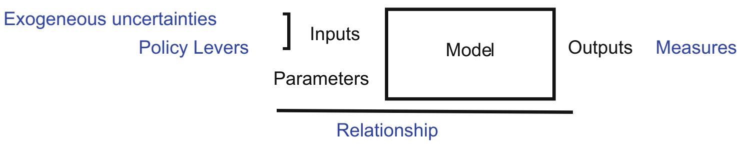


**Fig. 28.1** Diagram of a model

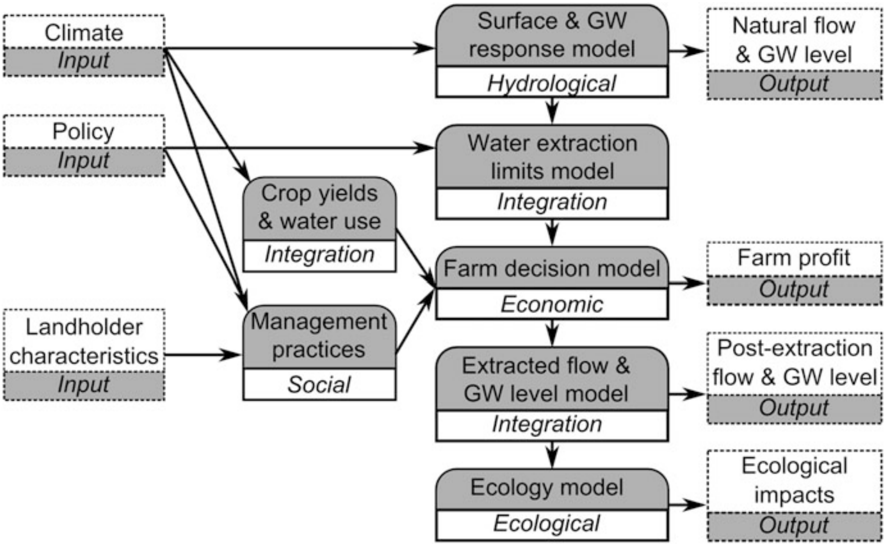


**Fig. 28.2** Diagram of an integrated model (Adapted from Jakeman et al. 2014)

dependent economic and ecological systems (Jakeman et al. 2014). At the core of the integrated model is the hydrological component. The hydrological model takes historical climate data, or future climate projections, as inputs and produces estimates of natural surface-water flows and groundwater levels as outputs. A Water Extraction Limits Model uses the estimated water availability and selected water policy options to calculate the allocations available to landholders. A Farm Decision Model then calculates actual water usage and farm profit based on the pre-extraction water availability, crop characteristics and the modelled decision-making behaviour of landholders. The landholder behaviour is simulated by a social model, which considers levels of compliance and adoption of various land management practices. Finally, the model uses post-extraction surface water flows and groundwater levels to assess ecological impacts.

Problem definitions may however be uncertain. They may be affected by constraints on the modelling exercise, such as on cost, time, availability and quality of data and expertise, not all of which may be apparent from the start. Different users may have different conflicting objectives or contradictory understandings of the problem (Brugnach et al. 2008). There may be linguistic ambiguity, with multiple conflicting interpretations of a statement, particularly where people of different professional or disciplinary backgrounds are involved. Even if a problem statement seems qualitatively quite clear, it may be difficult to translate it into quantitative terms. For example, a groundwater well may be considered unusable or “dry” before the bore itself becomes dry (e.g. if the water level falls below the pump intake or if the remaining saturated thickness is insufficient to meet a water need). Moreover, as more is learned, the predictions required may also evolve. Preliminary results might show that other scenarios or policies need to be considered, or it might become apparent that an accurate prediction is simply not possible given the available knowledge, but alternative predictions might still be possible.

In each case, failure to address uncertainty may result in useless predictions. We will limit our discussion of uncertainty in problem definition to a few basic principles:

- Modellers and stakeholders need to work together to define a problem, in a manner cognisant of the uncertainty involved. All parties should avoid oversimplification of defining the problem; guidelines on this issue are available elsewhere (e.g. Johnson 2008; Voinov and Bousquet 2010)
- Modellers and stakeholders need to actively seek out different perspectives of the problem. Casting a wide net for views will help ensure that they encompass not only those views important now, but also those that may become important later. This may involve drawing on multidisciplinary teams, considering different parts of a system or seeking out contrasting world views (van Asselt and Rotmans 2002). For example, creating policy and administering may have different requirements. Modelling techniques allow for multiple objectives to be included (e.g. Reed and Minsker 2004; Mantoglou and Kourakos 2007), so it is better to avoid narrowing down prematurely.
- Be prepared to iterate – do not expect it to be correct the first time. Even as the modelling exercise progresses, remain open to the potential for the problem definition and conceptual models to change dramatically (Bredehoeft 2005). This corresponds to a Bayesian view of the world, wherein data are used to progressively update prior understanding.

In traditional management literature, decision processes are considered to have three main stages: identifying a problem, developing possible courses of action, and selecting a course of action (Janssen 1992). However, it is also possible that in highly complex and deeply uncertain problems, the definition of the problem may be dependent on one's idea for solving it (Rittel and Webber 1973). Feasible objectives of groundwater management can be limited by practical constraints and uncertainties in how a system will respond to different management interventions. For example, the objective of groundwater management can be restoring groundwater storage to a specified level, or improving groundwater storage relative to the current level. We may find the uncertainty is too high to allow us to predict the actual groundwater storage, but we can predict the direction of change (e.g. improve from current) with higher level of certainty. This finding may trigger us to reconsider what type of management objectives are likely to be achievable given uncertainties and thus what indicators/predictions we want to include in the models.

Therefore, in the context of modelling for decision making, an iterative discovery method designed for co-development of management targets (reflected by the indicators/predictions and model produces) and interventions (reflected by the drivers and scenarios used in the model) can be useful for exploring feasible

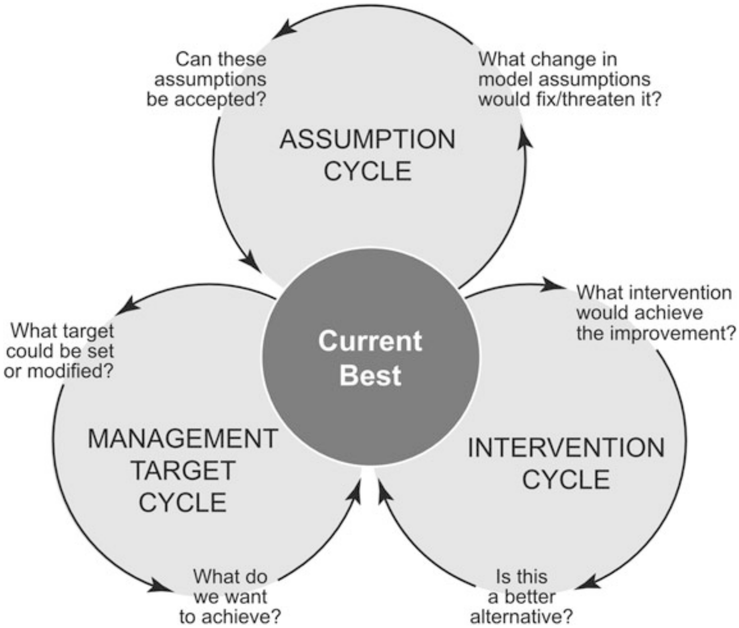


**Fig. 28.3** The iterative discovery method. Starting from the current best scenario, potential desirable and undesirable outcomes are identified which prompt the three cycles (assumption, intervention and management target) in order to identify achievable and specific management targets and alternative interventions under uncertainty (From Fu et al. 2015)

management targets under deep uncertainties (Fu et al. 2015). As shown in Fig. 28.3, the method starts by evaluating a scenario describing the current best solution, for instance the current groundwater policy. Using visualizations of the solution and its impacts, the user is prompted to identify desirable or undesirable outcomes of the current best scenario. This provides the starting point for three cycles, focusing on model assumptions, alternative groundwater management interventions, and management targets such as maintaining or improving or restoring groundwater storage or the health of groundwater dependent ecosystems. The outcome of this method is a list of management targets that can and cannot be achieved, the potential interventions that correspond to these targets, and the assumptions and uncertainties associated with these interventions. These outcomes can then be used as inputs for trade-off or cost-benefit analysis of different interventions to select a suitable course of action.

This iterative discovery method highlights the importance of using models for capacity building in groundwater decision making under uncertainties. Rather than simply providing 'the' answer, the method and models are used to build a knowledge partnership between modelers and decision makers. This kind of method is therefore most useful to analysts preparing recommendations rather than decision makers receiving them.

## 28.3 Communicating and Using Predictions of Uncertain Outcomes

The selection of methods to explore uncertainty in predictions depends not just on the problem, but also on how predictions will be used. This in turn depends on *why* uncertainty needs to be addressed. Notwithstanding earlier discussion, there may be times when uncertainty does *not* need to be addressed in detail, such as if a wrong prediction will have no impact, impacts are entirely reversible at little cost, or adverse effects of omission can be effectively handled by other aspects of decision making such as engineering safety factors. By way of contrast, we can identify six reasons that uncertainty in predictions may need to be addressed:

(a) **Testing whether a conclusion may be wrong.** Model predictions form the basis for expecting a result – a conclusion that might later turn out to be wrong. In groundwater management terms, this might include that a project satisfies regulatory requirements on impacts to groundwater, that an extraction limit will prevent drawdown, or that a contaminant will not reach a well. The simplest means of communicating uncertainty in this case is to present a model(s) that cannot be rejected as implausible and in which a conclusion is not guaranteed, for example, describing a potential preferential flow pathway that would be consistent with collected data.

(b) **Identifying plans that are robust given uncertainty.** Predictions are frequently used within a planning or decision-making framework. Decision-making frameworks based on the concept of robustness aim to identify plans that perform well in a set of models that includes potential unfavourable conditions. The "min-max" and "min-max regret" optimisation methods find a single 'robust' solution that respectively provides the best performance in the worst model scenario, or minimises regret if the future turns out to be described by a different model than expected (Kouvelis and Yu 1997). Extensions to this concept include considering multiple solutions that are close to the best one or considering more than just the worst case (Kalai et al. 2012). Optimisation under uncertainty also includes a number of methods that use sets of models with names such as 'chance constraints', 'stochastic programming', and 'probabilistic ranking' (Wagner and Gorelick 1989; Gorelick 1990; Chan 1993; Morgan et al. 1993; Bayer et al. 2008). For example, Feyen and Gorelick (2004) ensure that water-table elevations in sensitive wetland areas are not excessively lowered by the withdrawal of groundwater by verifying that constraints on hydraulic head are satisfied by all model realisations in a 'stack' of alternative models. Communication of uncertainty consists of describing the characteristics of models over which a plan has been tested.

(c) **Identifying uncertain factors that have the greatest influence.** The field of sensitivity analysis examines "how the variation in the output of a model ... can be apportioned ... to different sources of variation" (Saltelli et al. 2004;

Matott et al. 2009; Saltelli and Annoni 2010). This is typically done by comparing outputs of large sets of alternative models with known differences and calculating statistics developed for the purpose (e.g. Doherty and Hunt 2009). This can help to identify dominant and insignificant drivers of system behaviour, e.g. comparing the effect of pumping in multiple wells on drawdown or comparing the effect of parameters on a prediction. Understanding uncertainty in dominant drivers will facilitate associated reductions in uncertainty in model outputs. Communication focuses on providing a ranking of factors, often with some quantitative measure of significance. Note that a ranking will likely itself be uncertain because estimates of uncertainty are themselves uncertain, in which case it may be necessary to test whether the resulting conclusion may be wrong (See item *a* in this list).

(d) **Prompting changes to models or knowledge.** Model uncertainty is related to a model's limitations. Understanding the source and nature of that uncertainty can help improve modelling. This includes lack of knowledge, variability and contradiction (Refsgaard et al. 2007; Brugnach et al. 2008; Guillaume et al. 2012). Identifying a knowledge gap may prompt changes that could reduce uncertainty in predictions, such as collecting additional monitoring data. Not all uncertainties are equal – identifying important sources of uncertainty and knowledge gaps in models helps to prioritise research efforts (Fu and Guillaume 2014). Understanding the causes of variability may allow them to be explicitly modelled. Identifying the existence of contradictory views may allow the design of experiments to resolve the debate. For example: model construction itself is a means of dealing with uncertainty, as "the model-construction process organizes and formalizes potential conceptual models of a ground water system" (Hunt and Welter 2010). The field of identifiability analysis aims "to expose inadequacies in the data or suggest improvements in the model structure" (Matott et al. 2009). Data acquisition planning aims to inform what data should be collected (Beven 1993; James and Gorelick 1994; Dausman et al. 2010; Fienen et al. 2010, 2011). Each of these involves exploring and improving the state of inherently imperfect models. Communication of uncertain predictions focuses on its implications for later analyses, or on helping to justify why changes to a model have been made.

(e) **Providing quantitative estimates of uncertainty to other users.** The 'need' to provide an estimate of uncertainty is among the most commonly cited reason for using techniques to explore uncertainty in predictions. As the previous four points indicate, the need reflects a larger context, where uncertainty is a means to a decision-making end. In many cases, information about uncertainty can be communicated and used without necessarily expressing it in quantitative form. A more quantitative characterization of uncertainty may however be used in other processes, such as for risk management and decision theory (Freeze et al. 1990), and may be required by law in some countries as

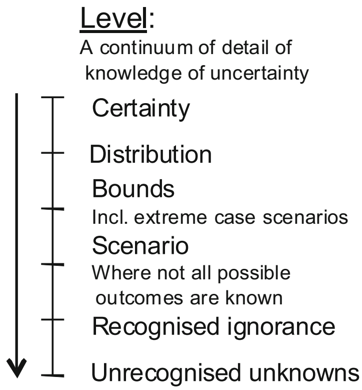


**Fig. 28.4** Levels of detail to represent quantitative estimates of uncertainty (Modified after Walker et al. 2003; Guillaume et al. 2010)

part of cost-benefit analyses or impact assessments. It may also be necessary to pass uncertainty information on to users without knowing how they will use it. In these cases, it is considered good practice to present "the modeller's estimate of the representative uncertainty given what is known about the system, the type of prediction(s), and the modeller's experience with the model and model calibration" (Hunt 2012).

Estimates of uncertainty can be represented at various levels of detail, as illustrated in Fig. 28.4 (Walker et al. 2003; Guillaume et al. 2010). For a given source of uncertainty, there may only be enough information to represent it as bounds or scenarios, rather than probabilities. For example, it might be more appropriate to use best-case and worst-case scenarios (Renard 2007; Paté-Cornell 1996). Where there are many sources of uncertainty, they may need to be represented at multiple different levels, for example variability of rainfall as a distribution, future prices of irrigated crops as bounds, and possible groundwater policies or irrigator pumping patterns as scenarios (Guillaume et al. 2012). Uncertainties that are known at a high level of detail can also be represented at lower levels of detail. For example, probabilities can be represented not just as probability distributions or cumulative distribution functions, but also using means and standard-deviations, confidence intervals and an ensemble of samples from a distribution.

In all cases, the consumers of the uncertainty estimate become the primary focus of how best to relate estimates to others. A groundwater scientist cannot expect that those needing to use the estimates will understand the academic terms and metrics (Hunt 2012). Therefore, translation of estimates into formats of direct use to the decision-making process should be used when possible (e.g. Hunt et al. 2001). Care needs to be taken when communicating estimates of uncertainty, particularly in the case of probabilities. Interpretation of probabilities tends to be biased, such that it is better to communicate them as frequencies, even if the probability refers to the likelihood of a one-off event (Anderson 1998). Rather than providing tables of probabilities, they may be better visualised (Barnett et al. 2012) by using maps or graphs. Where possible, expressing probability with its consequence allows it to be interpreted in terms of risk, reliability or probabilities of exceedance (Paté-Cornell 1996), and therefore provides a closer tie to its implications. Crucially, because of the potential for unrecognised unknowns, presentation of uncertain predictions should avoid stating uncertainty estimates in isolation. It is preferable to instead list the specific aspects of uncertainty that have been considered in producing an estimate, with the understanding that some may have been overlooked (Hunt and Welter 2010; Roy 2010; Guillaume et al. 2012). There is an extensive literature on the presentation and interpretation of uncertainty estimates (Wardekker et al. 2008; Kloprogge et al. 2007), even in the case of scenarios (Alcamo 2008).

(f) **Passing on qualitative information about uncertainty.** In a strict theoretical sense, the presence of unknowable model structure error means that true uncertainty cannot be characterized (e.g. Beven 2009), and true quantitative estimates are unattainable. Moreover, where a prediction is used only as a scenario to prompt discussion, a qualitative approach may be sufficient. The emphasis in this case may be on how the prediction was produced, and the limitations involved in doing so. For example, modelling of limits to growth was deliberately aimed to open a debate (Meadows et al. 1972), and uncertainty primarily needs to be addressed to convince the audience to take the arguments made by the model seriously. One way of approaching this is through quality assurance of the modelling process and its constituent assumptions (Refsgaard et al. 2005; Guillaume 2011). Another is to include qualitative judgements about the information and how it is produced (Funtowicz and Ravetz 1990; Kloprogge et al. 2011; Van Der Sluijs et al. 2005).

## 28.4 Methods for Generating Alternative Models

The preceding section described multiple ways of using uncertainty information. Alternative models often form an important construct within them for expressing uncertainty, where the uncertainty is represented by using a combination, or ensemble, of model realisations. Each model realisation can consist of different parameter values, inputs and/or model structures, as described in Fig. 28.1. The remainder of this chapter briefly presents a variety of methods for generating alternative models. Each section describes how the method produces models and key underlying assumptions with reference to examples. The methods are summarised in Box 28.1.

**Box 28.1 Types of Methods for Generating Alternative Models**

28.4.1 Models with different input values

28.4.2 Models with different 'structures'

- Models with different conceptualisations
- Models with different mathematical and computational implementations

28.4.3 Models with different parameter values

- Geostatistics: models satisfying statistical properties of data
- Parameter estimation: sampling models that fit data
- Multi-objective parameter estimation: sampling models that fit contrasting data
- Hypothesis testing: searching for models that fit data and satisfy a hypothesis
- Sensitivity analysis: selecting models to understand influence of drivers

28.4.4 Models to anticipate surprise

- Models to support adaptation
- Models that explore the unknown

### 28.4.1 Models with Different Input Values

Model results depend on the inputs of sources, sinks and system properties and initial and boundary conditions. Hydraulic heads are given as initial and boundary conditions at the start of the modelled period and at boundaries of the modelled aquifer, such as water levels in lakes, rivers or the ocean. Flows are given as conditions to capture inflows or outflows, whether above ground (e.g. pumping, streamflow or rainfall), or under-ground to and from outside the model area (e.g. regional groundwater flow).

Inputs and boundary conditions are approximate, can be expected to contain errors, and can be expected to change over time. Alternative models can therefore be created by changing the values of these inputs. We give some examples, but any model input could be altered. Values can be set based on expectations in the future (e.g. sea level rise, development of irrigation). Values can be randomly sampled from a feasible range of distribution in what is referred to as a Monte Carlo procedure. To capture historical variability, observations can be sampled from existing time series (e.g. Guillaume et al. 2012). Time series can be generated by using a statistical model, for example a weather generator. Outputs can be used from other studies, for example climate scenarios. Groundwater models can also be coupled or integrated with other models, such as ones that models surface water flows and levels (e.g. Graham and Butts 2005; Kollet and Maxwell 2006; Brunner and Simmons 2012), water flows and temperature (Hunt et al. 2013), or irrigator decision making and pumping (Hanson et al. 2010; Guillaume et al. 2012). Corrections to time series can also be made by using parameters that can be estimated along with other parts of the model (Vrugt et al. 2008).

These methods assume that it is sufficiently easy to modify the data used in the modelling software, and that input scenarios chosen are meaningful. It is not cost-effective or useful to produce many scenarios unless there is a clear way of summarising and understanding them, whether as a statistical distribution of a phenomenon, or as standalone scenarios.

### 28.4.2 Models with Different 'Structures'

As discussed with reference to Fig. 28.1, modellers distinguish the model proper from its parameters and inputs. The model proper is referred to as its 'structure', and can differ both in how it is conceptualised, which processes are included or excluded, and how it is implemented in mathematical and computational terms (Gupta et al. 2012).

#### 28.4.2.1 Models with Different Conceptualisations

A model's conceptualisation includes both its physical structure, such as the layout of an aquifer or catchment, and process structure, including recharge, aquifer flow and discharge mechanisms (Gupta et al. 2012). In groundwater flow modelling, physical structure tends to be a greater issue because the subsurface environment is observed by sampling, which is necessarily incomplete. Surprises in conceptualisation of the physical structure have included (Bredehoeft 2005) flow of brine within salt, faster flow through unknown factures, lack of evidence of whether a fault is or is not permeable, and lack of understanding of the connection of surface and groundwater, at surface seeps or in river bed. Although groundwater processes are generally well understood, there may still be unanticipated recharge events, unexpected effects of land subsidence, and overlooked chemical reactions.

There are several ways to provide diverse model conceptualisations, each with their own assumptions. A simple approach is to use a set of models pre-determined by hydrogeologists and modellers. However, it cannot be assumed that it is possible to identify all possibilities (Bredehoeft 2005).

An alternative is to approach modelling iteratively, building on previous effort (Haitjema 1995, p. 245; Bredehoeft 2005). This involves starting from an initial simple model, then using a stepwise process to identify limitations and refining models to include additional processes or physical structures. For example, there might be changes in system properties such as subsidence due to potential changes in human operations, such that indirect factors that influence pumping (e.g. economic considerations) should be included as part of a model rather than as a separate input (e.g. Hanson et al. 2010).

Using all possible models can however be overwhelming. Professional judgement or statistical criteria (e.g. Singh et al. 2010) can be used to rank the models or filter some out to know where to focus. However, this risks eliminating models that might turn out to better represent the unknowable future, so it is worth treating such a decision as provisional, and keeping an open mind about returning to the models excluded.

Fortunately for decision-making, it is often not necessary for the model to describe precisely what is occurring in the groundwater system. Instead, conservative estimates can be used that can inform decisions regarding margins of safety (e.g. in Bredehoeft 1983; Tiedeman and Gorelick 1993). In practice, the best one can hope for is to identify models that bracket the true value (Doherty 2011), from which safety factors can be derived from model results even if deliberately over- and under-estimated (Guillaume et al. 2012). This approach however assumes that there is a known bad thing to avoid (Freeze et al. 1990) and costs of being overly conservative are formally recognized.

#### 28.4.2.2 Models with Different Computational Implementations

Modelling requires that conceptualisations be made explicit in mathematical and computational form. Achieving this level of precision typically requires additional assumptions, regarding spatial variability, equations and their computational solution (Gupta et al. 2012).

Most numerical groundwater models discretize space into piecewise-constant quantities with a nodal grid or mesh. This discretization process raises the question of appropriate scale, and how the trade-off of computational burden and model resolution is decided. It is also possible to vary the resolution, and to use a combination of fine and coarse resolutions (Mehl et al. 2006). No grid or set of elements will fully capture a conceptual model, so the trade-off is subjective in that a modeller and end-user have discretion to select a variety of alternative scales based on practical considerations, such as computational cost. The objective of the model is of primary importance; models used to determine regional trends in groundwater level will require a different resolution than those used to evaluate the local flow of a contaminant.

### 28.4.3 Models with Different Parameter Values

Parameters play a key role in easily generating alternative models. In the context of groundwater modelling, Bredehoeft (2005) observed that “in many cases hydrogeologists were not sufficiently informed to imagine what is the entire set of possible conceptual models.” In most cases, detailed properties of specific groundwater systems would be even more difficult to specify a priori. Instead, the modeller uses a general structure and defines parameters that when varied will encompass a variety of specific system properties. Multiple sets of parameter values can then be specified, or can be estimated or constrained by observations from the field. A strength of these methods is that hydrogeologists' knowledge of the broad scale system, and its effect on local scale properties that result, can be tested and evaluated; and vice-versa, hydrogeologists' knowledge can be used to evaluate the quality of data.

#### 28.4.3.1 Geostatistics: Models Satisfying Statistical Properties of Data

Geostatistics provides a systematic means of using statistical properties of observed spatial data to generate alternative conceptualisations of physical properties. It interpolates given data points while satisfying observed heterogeneity and connectivity, which is particularly important for flow of contaminants (Renard 2007).

Data are used to generate a statistical model of heterogeneity (Marsily et al. 2005). The statistical model is in most cases a 'variogram', which captures the probabilistic degree of dependence between any two points in space (Delhomme 1979). More advanced techniques, like multiple-point statistics, adopt a richer model of heterogeneity where the relationship between complex patterns of points is enclosed in a so called training image (e.g. Mariethoz and Caers 2014), complementing data with additional geological 'soft' knowledge (Strebelle 2002; Hu and Chugunova 2008; Meerschman et al. 2013). Both these geostatistical methods are stochastic, in the sense that once a model of heterogeneity is selected (variogram or training image), an infinite number of equally probable realisations of geological heterogeneity can be generated, allowing the exploration of the corresponding uncertainties.

Four realisations of a multiple-point statistics conditional simulation are illustrated in Fig. 28.5, together with observed data points. Noting that a sand channel is observed at points 2, 4 and 3, one might ask: what is the probability that the points are connected by the same sand channel? In three of the four realisations presented here the three points belong to the same sand channel (realisations #1, #2, and #4) while in the other (realisation #3) the point 2 belongs to a different sand channel. The statistical techniques by which these realisations are generated allow probabilities to be calculated, given a sufficiently large number of realisations, if necessary assumptions are satisfied.

Geostatistical methods require dedicated tools and training to be used. They can require significant computer time, and depending on complexity of the problem, not all physical relationships in structure can yet be captured by theory. In addition, the choice of the model of heterogeneity in itself represents a source of uncertainty, closely related to the conceptualization of the geological model (see Sect. 28.4.2.1). Notwithstanding these drawbacks, geostatistical techniques are expanding from the mining and oil industries into the groundwater sector as they represent an important tool to explore uncertainty-related problems.

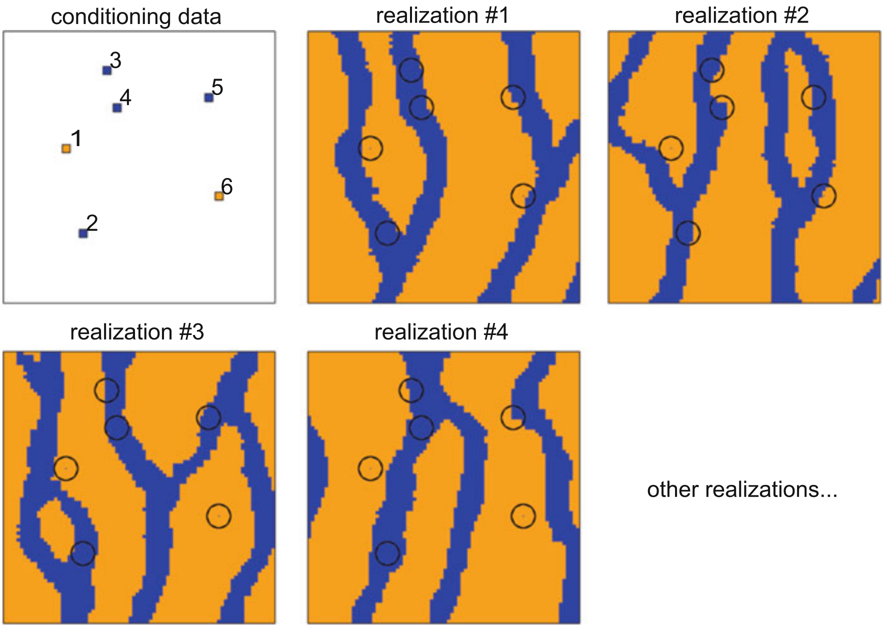


**Fig. 28.5** Conditioning data and four realizations of a multiple-point statistics simulation of a sand channel system (The training image used for the simulation is taken from Strebelle 2002)

#### 28.4.3.2 Parameter Estimation: Sampling Models That Fit Data

Parameters within a model can be easily manipulated, yielding a family of model realisations that can be explored, each with the same site geometry or structure (Barnett et al. 2012). So, for example, many models with different hydraulic conductivity and storativity in parts of an aquifer might fit the data relatively well, even if the overall structure of the aquifer is kept constant.

However, groundwater models typically carry sufficient resolution to represent hydraulic conductivity and storativity at a fine spatial and temporal scale, and it is often not possible to directly estimate parameters at that level of detail (Barnett et al. 2012), due to computational difficulties or expense of data collection. Instead, parameters are estimated for ‘zones’ or ‘pilot points’ from which all the more detailed parameters are calculated. The use of zones, also known as parameter lumping, involves subdividing the model based on geological boundaries or other reasons into regions that will be given the same hydraulic properties. Although this approach is conceptually simple, disadvantages include that it can be difficult to define such zones ahead of time, it may become apparent that geological properties do vary within a zone, and the abrupt changes in hydraulic properties at the edges of zones may not seem natural. Moreover, such a traditional zonation approach can reduce the effectiveness of the model to extract information from the field data in ways that cannot be quantified (Hunt et al. 2007; Doherty and Hunt 2010a). The pilot points approach involves setting parameter values at a fixed set of points and then interpolating across the model, using some of the geostatistical techniques described above (de Marsily 1978; Doherty 2003; Doherty et al. 2010a). This results in a more automated process generating a smooth variation of hydraulic properties.

Values of parameters can be estimated with a number of different approaches. A first aim is to identify a single model that best fits the observed data and the soft-knowledge of the system, to form a single construct for decision-making purposes. This initially involves a process of trial and error, trying different parameter values to progressively minimise the difference between the model outputs and data. This history matching is then typically automated using formal nonlinear regression methods, which automate the trial and error testing of parameters to minimise an 'objective function' that provides a measure of difference between model outputs and data. A variety of approaches are available for the design of the optimisation algorithm (Duan et al. 1992; Vrugt et al. 2003) parameterization approach (Doherty and Hunt 2010b), and selection of objective functions (Renard 2007; Schoups and Vrugt 2010; Bennett et al. 2013). Problems can prevent automated parameter estimation from consistently identifying a unique set of parameters (Sorooshian and Gupta 1983; Doherty and Hunt 2010b; Barnett et al. 2012). Most notably, all models simplify the world and leverage additional soft-knowledge to ensure the complexity of the model does not exceed the information available in the data (Jakeman and Hornberger 1993; Moore and Doherty 2005; Hunt et al. 2007; Barnett et al. 2012). This allows a single set of parameters to be identified, in a process known as regularisation (Moore and Doherty 2006), whether done ad hoc as in trial and error history matching, or with advanced algorithms (Hunt et al. 2007).

The second type of method does not seek to identify a single best parameter set but instead identifies realisations, or a set of models, that fit the data well enough by either statistical or less formal 'acceptable performance' criteria. Statistical criteria make assumptions about the distribution of errors (Schoups and Vrugt 2010), which, if correct, allow estimation of probability distributions of parameters. All models are in principle retained, but for a given output (e.g. hydraulic head at a point in time and space), models that yield extreme output values can be ignored. For example, by accepting that one in every 100 identical predictions could be wrong, a 99 % confidence interval can be calculated for the model output. Depending on the mathematical form of the model, 'linear methods' can be used to provide quick estimates (e.g. Doherty et al. 2010b). Even computationally more demanding techniques (e.g. when few parameters are used, Markov Chain Monte Carlo, Keating et al. 2010; Laloy and Vrugt 2012) are still approximate in that all estimates of uncertainty will be lacking in some regard (Barnett et al. 2012).

Approaches that use less formal 'acceptable performance' criteria can be quite diverse. Set membership methods identify parameters when the error in data is assumed to be bounded (Walter and Piet-Lahanier 1990). Generalised Likelihood Uncertainty Estimation (GLUE) extends this idea by defining limits of acceptability (Beven 2006, 2009) against which randomly sampled models are tested. Similarly Null Space Monte Carlo (Tonkin and Doherty 2009) uses theory about parameter estimation to randomly sample other parameters that satisfy a minimum performance requirement.

Many of these techniques that rely on random sampling may require a long time or many computers to run. This is particularly the case if the model is slow (takes longer than a few minutes), or if the method requires a large number of model runs (i.e. highly parameterized models). In these cases, it may be advantageous to use a 'surrogate model' (e.g. Keating et al. 2010; Doherty and Christensen 2011; Asher et al. 2015). A surrogate model uses a smaller number of model runs to then mathematically approximate a complex model using a simpler function. They therefore run faster and allow the more complex techniques to still be used.

#### 28.4.3.3 Multi-objective Parameter Estimation: Sampling Models That Fit Contrasting Data

The methods discussed in the preceding section can be extended to evaluate models against multiple types of data. As models are necessarily a simplification of reality, even if a model fits one type of data well, such as a local pumping test, it may not make accurate predictions of other outputs, such as regional flows. It is known that information about hydraulic head alone does not allow both recharge and transmissivity to be simultaneously estimated in some conditions (Haitjema 1995, 2006). It is therefore desirable to use a variety of data sources to determine in which alternative models we might have sufficient confidence (Kim et al. 1999; Schoups et al. 2005; Hunt et al. 2006; Renard 2007). Groundwater models in particular can potentially predict a number of different outputs, for which data can often be obtained. Flow data can be compared to predicted spring flows and leakage to and from a river. Temperature data can be compared to temperature resulting from mixing, e.g. of surface water and groundwater. Salinity or concentrations of some contaminants measured in the field can be compared to predicted concentrations of these substances. Use of tracer substances, either introduced or naturally occurring in the aquifer, can be compared to predicted flow paths, travel time and groundwater age (time since water entered the aquifer). Recent ecohydrological tracers such as viruses (e.g. Hunt et al. 2014) allow characterization of very short time of travel (<3 years) – ages not well characterized by traditional tracers.

It can also commonly occur that a model with a single parameter set is not able to simulate every prediction equally well. Instead, there is a trade-off between fitting different datasets that may or may not inform parameters important for prediction, and the prediction of interest. Therefore, it is recognized that multiple alternative models may be required to provide better predictions for when there is more than one prediction of interest (Moore and Doherty 2005).

Where the uncertainty in predictions is too great, models can be used to optimize data collection to cost-effectively reduce the uncertainty associated with a given prediction. For example, existing models can be used to estimate the effect of establishing a new monitoring borehole at particular locations (Dausman et al. 2010), though results may be affected by the existing assumptions in the models used (Fienen et al. 2011). New data collection often consists of extensions of existing head and flux monitoring networks, but can also encompass estimates of model parameters obtained from dedicated tests, such as from soil properties of geological drilling logs, lab tests and from measuring the response in groundwater level during aquifer tests (Illman et al. 2008).

#### 28.4.3.4 Hypothesis Testing: Searching for Models That Fit Data and Satisfy a Hypothesis

Alternative models can be explicitly identified that seek to test a hypothesis, rather than just focussing on fitting data, as is the case with all the previous methods to obtain models with different parameters. The idea is that communicating uncertainty will involve assessing the probability that something bad might happen (Freeze et al. 1990; Doherty 2011), for example, a contaminant reaches the drinking water well, or an ecosystem dies from lack of water. Knowing this ahead of time, we can explicitly search for a plausible model that might return such a prediction.

One approach is to define criteria by which to test whether alternative models are acceptable and to find the model that is closest to meeting the hypothesis, for example the model where the contaminant comes closest to the well, or the ecosystem has the least volume of water possible. This can be achieved by making conservative (yet defensible) simplifying assumptions, or by expressing the relevant criteria mathematically and using 'constrained optimisation' tools.

Yet, a priori determinations of what is plausible can artificially limit the range of alternative models evaluated. Moore et al. (2010) remove this limitation by expressing the problem as a trade-off of predicted value against fit to the observed data, which in turn allows the user to determine the level of acceptable uncertainty (Fig. 28.6). Rather than explicitly defining acceptable performance criteria, "Pareto front" graphs are drawn showing the intrinsic trade-off of the prediction reaching a

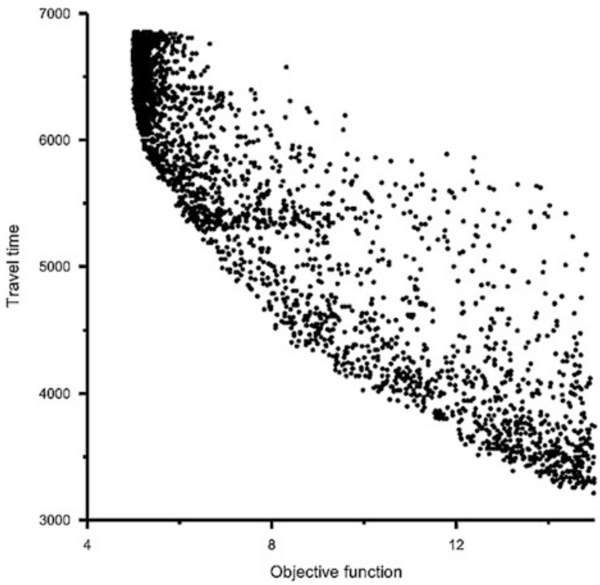


**Fig. 28.6** Pareto front defining trade-off between objective function (lower numbers indicate better fit) and predicted particle travel time (From Moore et al. 2010)

societally relevant threshold against the fit given by existing data for the system simulated. This makes it easy to relate uncertainty in terms of the prediction of interest, and can be constructed for multiple possible hypotheses and levels of acceptable criteria.

#### 28.4.3.5 Sensitivity Analysis: Selecting Models to Understand Influence of Drivers

Purpose-designed methods are also available where the aim is to efficiently identify uncertain drivers – those that have the greatest influence. These techniques are commonly referred to as 'sensitivity analysis' (Saltelli et al. 2004). These techniques may be based on local perturbations, where they only provide information about the specific model and set of parameters that is perturbed. Local sensitivity information may not reflect the sensitivity over the full range of plausible parameters. Therefore, sensitivity methods can also be global, which provides information about a broader sample of values the different sources of variation can take. Sensitivity analysis can either provide information about the effect of a factor keeping all others constant (Saltelli and Annoni 2010), or total effect of a factor with interactions with other factors (e.g. global sensitivity statistics such as Morris, Sobol, FAST).

### 28.4.4 Models to Anticipate Surprise

It is inevitable in all environmental modelling that there might be 'unknown unknowns' and therefore surprises (Bredehoeft 2005; Hunt and Welter 2010). Model structures are unlikely to serendipitously include processes or structures that modellers do not know exist. Models that are calibrated by using existing data are tuned to reflect processes that can be identified from that data. Predictions of flow of contaminants could be completely underestimated if fractures exist that were not explicitly incorporated into the model. We discuss two approaches to creating models that help deal with this surprise: models to support adaptation, and models to explore the unknown.

#### 28.4.4.1 Models to Support Adaptation

In principle, surprise can be dealt with by adaptive management. Rather than expecting modelling to anticipate all uncertainty, management plans remain open to change and plans for an iterative modelling and management process (Bredehoeft 2005). In the context of groundwater, models are still crucial to this process. Due to slow response times, when a change is detected, it may already be too late to do anything about it (Bredehoeft and Durbin 2009). Even if action is taken immediately, impacts may still worsen before they improve. It is therefore essential to try to anticipate the delays that might occur within a system. Model scenarios can also help to predict "sell-by" dates at which current plans might be expected to fail, to help plan adaptive pathways (Haasnoot et al. 2013). As discussed earlier, models can be used to help plan the monitoring needed to detect unexpected changes with an understanding of the time until impact. In parallel, model scenarios can be created to evaluate how future options might be curtailed as a consequence of short-term choices (Wong and Rosenhead 2000). Methods exist to allow model parameters to be efficiently updated given new data, and to detect when the data does not fit the current model (e.g. Cheng et al. 2011). This is particularly relevant where models are used operationally, such as in mine dewatering and water supply.

#### 28.4.4.2 Models to Explore the Unknown

Potential surprises can also be anticipated by placing fewer restrictions on what is considered possible, and using the model prediction as a discussion point. Considering a larger set of models helps inform adaptation by discussing "what we do if this situation did occur?" This is even possible if no data are available.

This can be thought of as vulnerability analysis, identifying model properties in which negative outcomes occur. For example, Nazemi et al. (2013) identify changes in climate that would result in water scarcity problems, deferring the judgement as to whether those climate changes could occur. Scenario discovery (Bryant and Lempert 2010) randomly samples a large number of parameters and then identifies the values of parameters for which the negative outcome might occur. Break-even analysis identifies models at tipping points, for example, the infiltration rate or hydraulic conductivity at which managed aquifer recharge using basin infiltration is uneconomical (Frey and Patil 2002). Similar techniques have been applied to identify the circumstances in which two management options are equivalent, i.e. the point at which a different option becomes superior (Ravalico et al. 2009).

## 28.5 Conclusions

This chapter discussed a variety of methods for generating alternative models in order to explore uncertainty in predictions that can be applied to integrated groundwater management. The methods used depend on how the problem is defined, resources available, and how it is intended that the predictions of uncertain outcomes are used. Although many of these methods require hydrogeological, mathematical and computational expertise, together they provide a broad toolbox for identifying a more encompassing view of what might happen. Stakeholders are more likely to be forewarned with a range of plausible alternatives that they may have to face, which, in turn, can facilitate better decision making.

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