CSIRO PUBLISHING

www.publish.csiro.au/journals/mfr *Marine and Freshwater Research*, 2010, **62**, 1–10

A new era in catchment management: integration of environmental flow assessment and freshwater conservation planning

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Abstract. Integrated water resources management offers an ideal platform for addressing the goals of freshwater conservation and climate change adaptation. Environmental flow assessment and systematic conservation planning have evolved separately in respective aquatic and terrestrial realms, and both are central to freshwater conservation and can inform integrated water resources management. Integrating these two approaches is mutually beneficial. Environmental flow assessment considers dynamic flow regimes, measuring social, economic and ecological costs of development scenarios. Conservation planning systematically produces different conservation scenarios that can be used in assessing these costs. Integration also presents opportunities to examine impacts of climate change on conservation of freshwater ecosystems. We review progress in environmental flow assessment and freshwater conservation planning, exploring the mutual benefits of integration and potential ways that this can be achieved. Integration can be accomplished by using freshwater conservation planning outputs to develop conservation scenarios for assessment against different scenarios, and by assessing the extent to which each scenario achieves conservation targets. New tools that maximise complementarity by achieving conservation and flow targets simultaneously should also be developed.

Additional keywords: biodiversity, climate-change adaptation, integrated catchment management, integrated water resources management.

Introduction

Sustainable development of water resources is critical if society is to derive long-term benefits from freshwater ecosystems. Integrated water resources management is a widely accepted approach to support the sustainable development of water resources, coordinating the activities of multiple sectors in developing and managing water and related resources in an equitable manner, while conserving and restoring freshwater ecosystems (Global Water Partnership 2000). It supports a catchment-level approach to water resources management, which is important given the connected nature of most freshwater ecosystems, which makes them susceptible to impacts from upstream, downstream and surrounding landscapes.

Over the past 30 years, environmental flow assessment has evolved into a tool to inform integrated water resources management, linking changes in the natural flow regime from proposed development to ecological responses and condition (Poff *et al.* 2010). Resulting scenarios of catchment development can be used to assess the ecological, social and economic consequences (Dollar *et al.* 2010). They also help formulate a long-term vision for the catchment and ultimately guide subsequent activities. At the same time, systematic conservation planning was evolving as a planning and assessment tool in conservation science. With its overarching goal of planning for the long-term persistence of biodiversity, systematic conservation planning offers a practical tool, identifying areas of relative biodiversity importance. The decline in freshwater biodiversity (Ricciardi and Rasmussen 1999; WWF 2004) and numerous calls for its conservation (Abell 2002; Dunn 2003; Nilsson *et al.* 2005) have increased application of systematic conservation planning approaches to freshwater systems (Nel *et al.* 2009; Linke *et al.* 2010). These freshwater conservation planning tools identify priority systems for freshwater conservation, providing a strategic conservation framework across entire regions. This strategic conservation framework is lacking in most integrated water resources management programs (Gilman *et al.* 2004).

 10.1071/MF09318 1323-1650/10/S10001

Environmental flow assessment and freshwater conservation planning have evolved separately, with the former originating from the water sector (Tharme 2003), and the latter primarily emerging from the terrestrial realm (Margules and Pressey 2000). They should complement each other in the conservation of freshwater ecosystems. They both inform integrated water resources management, share similar principles, include environmental sustainability, allocate limited resources efficiently, are scientifically defensible and engage stakeholders (Richter *et al.* 2006; Nel *et al.* 2009). The science and scope of the tools has reached a point of maturity and practicality for meaningful integration to enhance the respective tools and ultimately their use in water resources management. The present paper provides a starting point for integration. We begin by outlining the key objectives of environmental flow assessment and conservation planning and how they have evolved. We then describe the mutual benefits that could be realised through integration, recognising the potential of such integration for planning under climate change. Finally, we explore implementation of integration, highlighting challenges.

Environmental flow assessment

Removal of water from rivers, and other changes to their flow regimes, invariably result in a loss of ecosystem function and resilience (Bunn and Arthington 2002). The greater the divergence from a natural flow regime – in terms of volume and timing – the greater the ecosystem changes (Poff and Zimmerman 2010). Environmental flow assessments are designed to evaluate how much a river ecosystem changes with alterations to its natural flow regime (Tharme 2003). Resultant environmental-flow recommendations describe the flow regime necessary to sustain key ecological and societal values. Today, environmental flow assessments focus on the following two main contexts: management and/or restoration of single systems or sites, and integrated water-resources assessment and planning, usually at the catchment scale. The present paper focuses on the latter.

Environmental flow assessments began in the 1940s in the USA, and gained momentum after the 1970s (Tharme 2003). Since then, a vast number of approaches have been developed to assess environmental flows (Tharme 2003; Arthington *et al.* 2010). Early environmental flow assessments provided scientific input to water-use decisions on the needs of rivers, which were (and often still are) based solely on engineering solutions. They typically had narrow objectives, focussing on the minimum flow for single, valued species (usually fish). Over the past two decades, approaches have evolved to become more appropriate for informing integrated water resources management, with significant adaptations. First, instead of using individual species or a specific ecosystem service, the focus has expanded to more 'holistic' approaches that encompass the whole aquatic ecosystem, including the riparian zones, floodplain and estuary. This more comprehensively addresses the set of social and economic benefits derived from the ecosystem, which is especially important for the livelihoods of associated rural communities (King and Brown 2010).

Second, there was a shift from single, prescriptive flow-regime recommendations to producing scenarios (Brown and King 2010). Scenario construction is aided by establishing flow-response relationships that allow prediction of response indicators to increasing changes in the flow regime. Over the years, the largely biophysical response indicators (e.g. bank erosion, water chemistry, rare species, riparian forests) have also been supplemented with indicators relevant to social (e.g. livestock health, recreational sites, transport) and economic (e.g. employment, gross domestic product) consequences (Dollar *et al.* 2010; King and Brown 2010). Social, economic and ecological trade-offs can then be assessed in a stakeholder-driven negotiation process.

Third, conceptual frameworks and tools have recently been developed for environmental flow assessments across large geographical scales (e.g. catchments or bioregions with regulated and unregulated rivers). These regional approaches allow for configuration of rivers at different levels of ecological condition (natural to highly modified), depending on societal aspirations. Trade-offs at the catchment level allow identification of configurations that optimise social, economic and ecological outcomes (King and Brown 2010). The expanded geographical scale of these assessments necessitates approaches that can be applied with limited existing data, achieved by combining empirical measurements with modelled data and expert opinion. A key premise is that rivers in a hydrogeomorphological class (e.g. stable groundwater streams, or seasonally pulsed snowmelt streams) have similar flow-ecological responses that can be applied across the class, without requiring data for all rivers (Arthington *et al.* 2006; Kennard *et al.* 2010). Because of associated uncertainties, implementation of environmental flows requires an effective adaptive management framework with monitoring that tests ecological responses to management for further refinement (King and Brown 2010; Poff *et al.* 2010).

These advances provide opportunities for incorporating quantitative environmental-flow recommendations into integrated water resources management. Several decision-support frameworks and tools exist for scenario-based regional environmental flow assessment (Tharme 2003; Richter *et al.* 2006; Dollar *et al.* 2010; Poff *et al.* 2010). There are nine key steps (Fig. 1), as follows: (1) identification of 'analysis nodes' where water-management decisions are needed or anticipated; (2) collation of flow data for each node (long-term gauged daily data or hydrological modelling); (3) classification of rivers expected to respond similarly to flow alterations; (4) determination of deviation of current from natural flow regime at each node; (5) selection of ecological-, social- and economic-response indicators and relevant data; (6) construction of flow–response relationships of each river type and changes in flow-regime metrics; (7) development of a range of potential water-use scenarios and their impact on the flow regime; (8) evaluation of social, economic and ecological consequences of each scenario (relative to present day) with stakeholders to arrive at a consensus scenario; and (9) monitoring of ecological response to management and refinement where necessary.

Many flow–response relationships can be produced reflecting the response of each river type for each response indicator to changes in each of the flow-regime metrics. This information needs to be synthesised so that it can be understood by non-technical stakeholders. While acknowledging that it is ideal to use all relationships, Poff *et al.* (2010) proposed a new

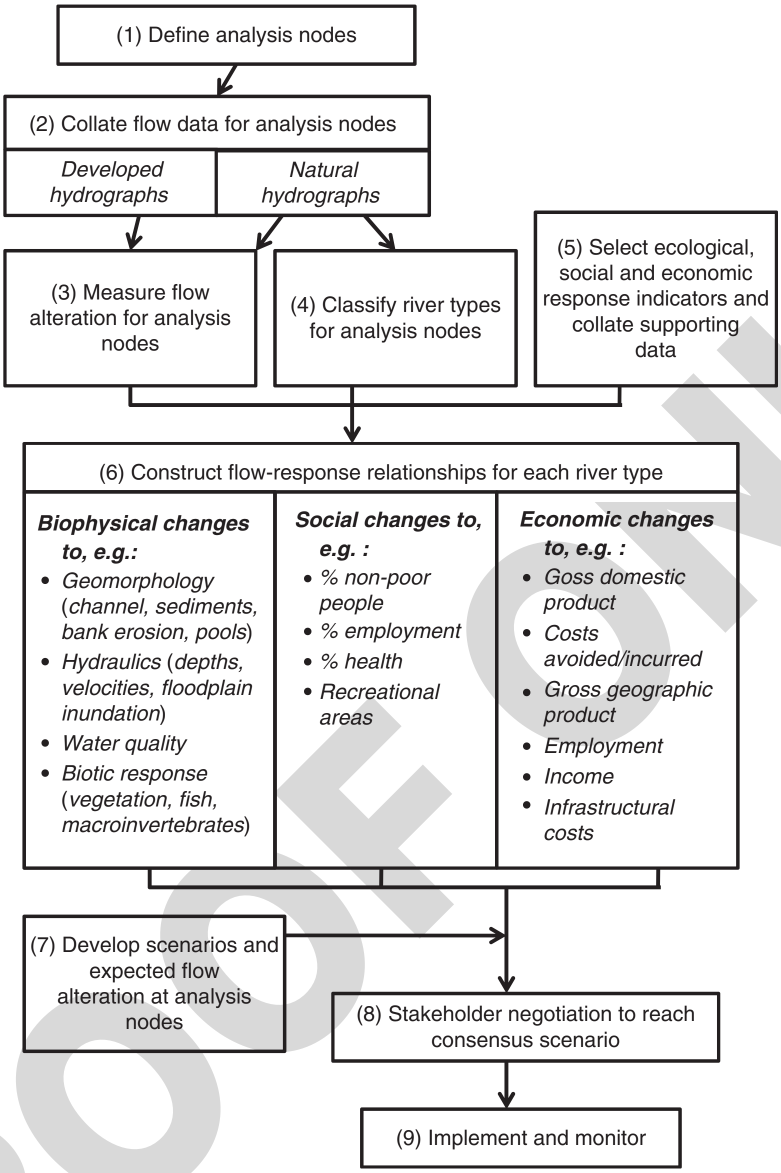

Fig. 1. Nine key steps in regional, holistic environmental flow assessments for evaluating the ecological, social and economic consequences of different development scenarios relative to current conditions. The process should be embedded within a stakeholder-driven implementation process linked to an adaptive management cycle (feedback loops omitted).

framework (ecological limits of hydrologic alteration, ELOHA) within which a parsimonious suite of flow–response relationships are extracted for evaluation. This selection would depend on the sensitivity of the river type to a particular flow alteration, the ability to manage a particular flow metric, the certainty of the flow–response predictions, and the relevance to stakeholders. The downstream response to imposed flow transformations (DRIFT) tool (Brown and Joubert 2003) develops summary ecological-condition categories of the overall deviation of the current from the natural condition. For each scenario, ratings of flow changes for each response indicator are combined using a set of pragmatic rules to derive overall ecological-condition categories relative to natural condition (Table 1). These overall ecological-condition categories can be embedded in policy processes (e.g. South Africa water policy) and are easy to communicate to stakeholders. They also considerably simplify evaluation of the social and economic consequences, linked to overall ecological condition rather than to each flow metric (Dollar *et al.* 2010). Determining the overall ecological condition is problematic because factors other than flow (e.g. water quality or habitat structure) affect ecological condition (Ormerod *et al.* 2010), and distinct boundaries of flow alteration, and hence ecological condition, are difficult to define. Recognising this, detailed flow–response relationships within DRIFT scenarios can also be accessed for further evaluation.

Table 1. Ecological-condition categories (after Kleynhans 2000) used in the downstream response to imposed flow transformations (DRIFT) environmental flow assessment tool (Brown and Joubert 2003) to describe the deviation of ecosystems from their natural condition

Ecological-condition category	Description
A	Unmodified, natural.
B	Largely natural with few modifications. A small change in natural habitats and biota may have taken place, although the ecosystem functions are basically unchanged.
C	Moderately modified. A loss and change of natural habitat and biota have occurred, whereas basic ecosystem functions are still predominantly unchanged.
D	Largely modified. A large loss of natural habitat, biota and basic ecosystem functions has occurred.
E	Seriously modified. The loss of natural habitat, biota and basic ecosystem functions is extensive.
F	Critically/extremely modified. Modifications have reached a critical level and the system has been modified completely, with an almost complete loss of natural habitat and biota. In the worst instances, the basic ecosystem functions have been destroyed and the changes are irreversible.

Scenario evaluation ideally incorporates risk assessment, in which stakeholders identify thresholds beyond which ecological, social or economic systems reflect unacceptable change (Poff *et al.* 2010). Thus, the environmental flow becomes the flow regime described by the consensus scenario for each river. These can be converted into a desired ecological-condition category for each river (see Table 1; Dollar *et al.* 2010).

Systematic conservation planning

Systematic conservation planning originally had a very narrow focus of locating formal protected areas. It grew from the realisation that the world's protected-area systems were biased in biodiversity representation, most commonly favouring areas of low economic potential (Pressey 1994). Early conservation planning focussed on systematic methods for biodiversity representation in protected areas and strategic use of conservation resources (Kirkpatrick 1983). This approach had limited applicability to connected ecological units such as freshwater ecosystems (Dunn 2003). Even in terrestrial settings, there were limitations. First, ecosystems within protected areas were essentially 'locked away' from human use, thus inadequately accounting for compatibility between human use and conservation (Richter *et al.* 2003). Second, protected areas alone are insufficient at conserving the variety of biodiversity. They should be cornerstones of biodiversity conservation, supplemented with other conservation strategies (Margules and Pressey 2000). Third, areas selected for representation are often not adequate for supporting natural processes that are essential for the long-term persistence of biodiversity, such as large migration corridors (Balmford *et al.* 1998).

To address these problems, the scope of systematic conservation planning has expanded to explicitly include other conservation mechanisms that acknowledge the needs of both people and ecosystems, ranging from highly restrictive (e.g. protected areas) to less restrictive (e.g. conservation easements, land stewardship) (Margules and Pressey 2000). In addition to locating the most strategic protected areas, systematic conservation planning is now a tool that informs land-use decision-making. Natural processes vital to the persistence of biodiversity are also incorporated (Pressey *et al.* 2007; Klein *et al.* 2009*a*). Design criteria, such as connectivity, are therefore explicitly recognised as a critical component of systematic conservation planning (Balmford *et al.* 1998).

The key objectives of systematic conservation planning are to represent biodiversity and plan for its persistence by using limited conservation resources efficiently (Margules and Sarkar 2007). It should also be embedded in stakeholder-driven implementation (Knight *et al.* 2006). There are six key steps in most current systematic conservation planning (Fig. 2). They include (1) delineation of 'planning units' for assessment, (2) selecting surrogate measures for biodiversity pattern and process, and mapping their distribution, (3) setting quantitative conservation targets for representation and persistence, (4) setting constraints (e.g. management costs of conservation actions or economic impact on stakeholders), (5) selecting planning units that achieve the conservation targets for all biodiversity most efficiently and (6) implementing, monitoring and adaptively managing conservation actions. This last step can be automated by using complementarity-based conservation planning tools (Sarkar *et al.* 2006). Complementarity promotes efficiency in the number or area of selected planning units by choosing not just the planning unit with the most biodiversity features, but rather the one that contains the most unrepresented features (Kirkpatrick 1983; Pressey 1994). Areas are chosen to complement, not duplicate, each other's biodiversity. The incorporation of social and economic constraints of each planning unit also improves the cost efficiency of conservation action, although this has only recently been done in terrestrial and marine settings (Carwardine *et al.* 2008; Klein *et al.* 2008) and not in freshwater settings.

Two maps can be generated from most generic conservation planning tools. One is a relative index for each planning unit of the required likelihood to meet conservation targets ('irreplaceability map'); planning units with high irreplaceability have few options for replacement, whereas planning units with low irreplaceability can be replaced (Ferrier *et al.* 2000). The second map displays the most efficient spatial solution for achieving conservation targets at the lowest possible cost ('minimum set' map). These guide conservation planners in the formulation of a conservation framework for the catchment with stakeholders and regional conservation experts.

As systematic conservation planning has advanced, it has become conceptually suitable to planning for freshwater

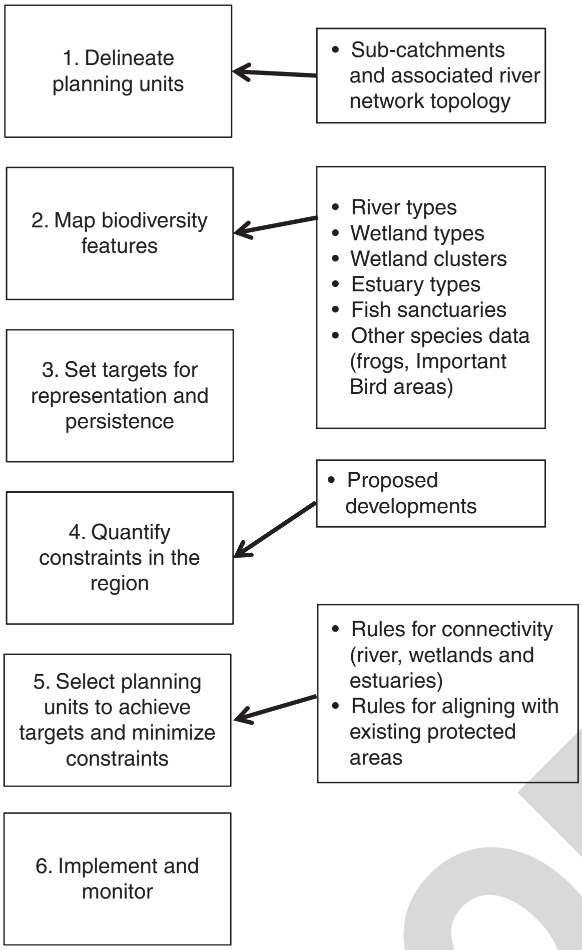

Fig. 2. Six key steps in freshwater conservation planning with examples of supporting data. Steps are inter-dependent and not necessarily unidirectional. The entire process should be embedded within a stakeholder-driven implementation process.

biodiversity (Nel *et al.* 2009; Linke *et al.* 2010). Freshwater conservation plans delineate subcatchments as planning units, incorporating the need to manage the water resource and its catchment (Lehner *et al.* 2006; Thieme *et al.* 2007). Many include the development of a river network topology (tree hierarchies) to assess the position of candidate subcatchments, relative to other upstream or downstream ones (Moilanen *et al.* 2008). This incorporates explicit longitudinal connectivity into complementarity-based conservation planning tools (Linke *et al.* 2007). Most plans also consider multiple-use zonation, with different levels of protection depending on the role of that particular subcatchment in achieving biodiversity goals (Thieme *et al.* 2007; Turak *et al.* 2010). Zones are broadly based on the concept of hierarchical protection in which freshwater focal areas are embedded within critical management zones, which, in turn, are embedded in catchment management zones (Abell *et al.* 2007). Freshwater focal areas largely represent natural examples of freshwater ecosystems and their associated biodiversity, and their use is likely to be restrictive. Use restrictions diminish in the latter two zones where the focus is largely on persistence.

Mutual benefits of integration

Systematic representation of freshwater biodiversity

Environmental flow assessments frequently incorporate at least one scenario addressing the conservation of important freshwater ecosystems. This is usually based on an index of ecological importance (Dollar *et al.* 2010) that scores each system by using criteria such as richness, rarity, endemism, importance for processes such as migration, naturalness, presence of protected areas, fishing value and ecotourism potential (Kleynhans 2000; Dunn 2003). This single relative index of ecological importance is used to assess ecological, social and economic impacts of different water-use scenarios.

Although the simplicity of the index of ecological importance is appealing, scoring approaches are conceptually flawed when identifying priority areas for conservation (Pressey and Nicholls 1989; Possingham *et al.* 2006). They under-represent the full suite of freshwater biodiversity because ecosystems with low overall scores are left out. They also ignore complementarity failing to maximise efficiency in achieving biodiversity goals. Modern freshwater conservation planning methods overcome these issues, producing a comprehensive solution that includes all conservation targets, while minimising impact on stakeholders (e.g. irrigators). Using freshwater conservation planning tools rather than scoring for ecological importance provides a more efficient, representative and scientifically defensible approach for identifying high-value conservation areas.

Incorporating flow-regime dynamics into conservation planning

The persistence of freshwater biodiversity is greatly influenced by hydrological connectivity along three spatial dimensions (longitudinal, lateral and vertical) and a temporal dimension, all linked to the natural flow regime (Pringle 2001). Freshwater conservation planning has focussed on incorporating the spatial dimensions of connectivity by identifying upstream, downstream and lateral linkages potentially affected by human impacts. The temporal dimension of the natural flow regime – the variability in magnitude, timing, frequency, duration, rate of change and predictability – has received only rudimentary consideration in freshwater conservation planning. The temporal dimension is usually captured by indices of alteration. These include disturbances that can be mapped as surrogate measures of likely flow-related changes (e.g. extent of impervious surface in the catchment, extent of irrigated agriculture, density of dams, levees and weirs). These are combined with surrogate measures of non-flow-related change to derive an index of overall ecological condition for rivers (Stein *et al.* 2002; Turak *et al.* 2010) and wetlands (Ausseil *et al.* 2010). These indices of ecological condition are used to set explicit rules in conservation planning tools. For example, if choices exist between two sites with similar biodiversity characteristics, the site with the best ecological condition is favoured (Turak *et al.* 2010). Ecological-condition indices are also used to formulate different conservation strategies for sites selected in the conservation portfolio; e.g. sites selected in a conservation plan that are in poor ecological condition may be highlighted as priority areas for restoration (Linke *et al.* 2007).

The methods of examining flow alteration in freshwater conservation planning are simplistic, and would benefit greatly from the understanding gained over three decades of environmental flow-assessment research on modelling flow-regime dynamics (King *et al.* 2003; Arthington *et al.* 2006; Poff *et al.* 2010). In addition, catchment-wide environmental flow-assessment recommendations deliver quantitative flow recommendations necessary to maintain different overall ecological conditions (Dollar *et al.* 2010), which can be used to develop guidelines that should accompany conservation planning products (Knight *et al.* 2006). This opportunity could be further exploited by ensuring that the environmental flow-assessment exercise includes ecological-response indicators that are directly relevant to the biodiversity measures used in the conservation planning exercise. Individual metrics of relevance for developing specific conservation-management activities can then be examined, as well as the index of overall ecological condition that summarises multiple ecological response and flow-alteration relationships.

Explicit scenario-based analysis of social and economic implications

Freshwater conservation planning has not yet explicitly considered the social and economic costs and benefits of conservation action. Incorporation of conservation costs greatly improved the alignment of biodiversity goals with social and economic objectives in terrestrial and marine settings (Carwardine *et al.* 2008; Klein *et al.* 2008, 2009*b*). For freshwater ecosystems, the process of examining conservation costs is complex because of the range of upstream, downstream and upland dependencies. Modern environmental flow assessment evaluates different water-use scenarios by explicit linking of flow changes (and consequent changes in ecological condition) to social and economic costs and benefits within this dependency framework (King and Brown 2010). Incorporation of such methods in freshwater conservation planning will improve the social and economic cost-efficiency of achieving biodiversity goals. Environmental flow assessment also assesses social and economic consequences for all systems in the catchment, not just the ones of high conservation value, highlighting the need to manage the catchment as an integrated whole.

Another key strength of environmental flow assessments in the context of integrated water resources management is objective assessment of social, economic and ecological consequences of flow changes from a range of different development scenarios, rather than prescription of a specific flow regime. This scenario-based approach empowers stakeholders to formulate their own vision for water resources in the catchment. With increasing availability of finer-scale climate-change scenarios (Saji *et al.* 2006), environmental flow assessment can also evaluate ecosystem responses to combined development and climate-change scenarios. Predicted changes in rainfall and temperature will have an impact on water resources; moreover, rainfall changes tend to be amplified in changes to the flow regime. For example, in modelling the changes to the variability of inter-annual flooding, some regions have shown a two- to five-fold amplification of change from rainfall to runoff (Schulze 2005); this has important implications in future planning around extreme events such as floods and droughts. Changes to the flow regime under various climate-change scenarios can be quantified (Schulze 2005; Palmer *et al.* 2008) and used in an environmental flow-assessment exercise to examine the likely impacts of development and conservation within a changing climate. Such proactive planning offers an ideal starting point for considering climate change-adaptation strategies within the context of water-resources development, but have yet to be explored.

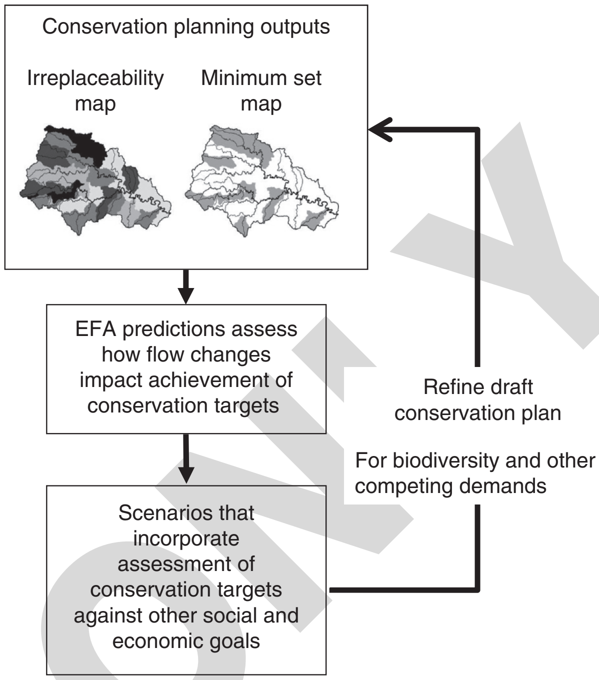

Fig. 3. Freshwater conservation planning could be used within existing environmental flow assessment frameworks, beginning with two maps of a catchment. These can then be used to assess achievement of conservation targets with flow-related changes. This provides explicit information for assessing trade-offs between conservation and development. The process can be iterative where conservation planning products are refined by using the conservation options available under the selected development scenario.

Options for integration

Conservation targets for scenarios

For each scenario assessed, the ecological condition (Table 1) resulting from flow-related change must not just be translated into social and economic consequences, but also evaluated against the achievement of conservation targets (Fig. 3). This makes the trade-offs between conservation and development explicit. Such assessment can be achieved in several ways. One way is to use a specific conservation scenario to compare achievement of conservation targets under current ecological conditions with conditions under proposed development. Such a single conservation scenario represents only one of many possible conservation solutions, and offers no information on other conservation options in the light of development options (Ferrier *et al.* 2000). An alternative is to use the irreplaceability of each subcatchment to produce the options available for conservation. Development in subcatchments with high irreplaceability values should be penalised, since these subcatchments are probably most needed to achieve conservation targets. The use of irreplaceability may address such prescription, but comes with the following disadvantages: selection of only highly irreplaceable subcatchments for conservation will not achieve all targets; and some conservation choices still need to be made between subcatchments with low irreplaceability. The extent to which development exhausts all options in subcatchments with low irreplaceability therefore also needs to be included in such an assessment.

Environmental flow assessments and freshwater conservation planning should be at comparable levels of resolution, ideally using the same planning units (subcatchments or stream segments). Each subcatchment requires an analysis node, with associated hydrology and flow–response relationships (Fig. 1). Hydrological data at subcatchment scales are lacking in most parts of the world, although this obstacle can be addressed through hydrologic simulation models of rainfall-runoff and other catchment processes (Kennen *et al.* 2008). If no data are available, river-type classification can then be used to extrapolate flow responses across to analysis nodes belonging to the same river type (Arthington *et al.* 2006; Kennard *et al.* 2010), to quantify different configurations of desired ecological condition relative to water use in the catchment.

The way freshwater conservation targets are set and communicated is a potential obstacle. Scientifically defensible targets are usually defined on the basis of specific requirements for the persistence of biodiversity (Carwardine *et al.* 2009); however, minimum population sizes or minimum habitat requirements for the persistence of most freshwater species are not known (Abell 2002). Freshwater conservation planning commonly sets socio-political targets, reflecting societal aspirations. The 20% target for all aquatic biodiversity is commonly adopted by freshwater conservation planning (IUCN 1991). Such socio-political targets of simple percentages lack scientific defensibility and may not result in the persistence of biodiversity (Svancara *et al.* 2005). Nevertheless, they reflect an aspiration for equity in representing biodiversity. Socio-political targets are useful for measuring progress towards conservation and informing integrated water resources management, particularly if they symbolise a societal aspiration endorsed by policy and effectively communicated to stakeholders (Roux *et al.* 2009). However, they are preliminary and should be updated with new ecological information and changing social values of conservation.

Integration of conservation planning and flow targets

A simple option for integrating freshwater conservation planning and environmental flow assessment would be to complete a freshwater conservation plan for the catchment, producing an irreplaceability and minimum-set map. The minimum-set map would form the conservation scenario for evaluation against development scenarios (Fig. 3). Because there are many possible configurations to achieve conservation targets in the light of development options, a better solution to integration would be to maximise biodiversity and flow targets simultaneously, while minimising social and economic costs.

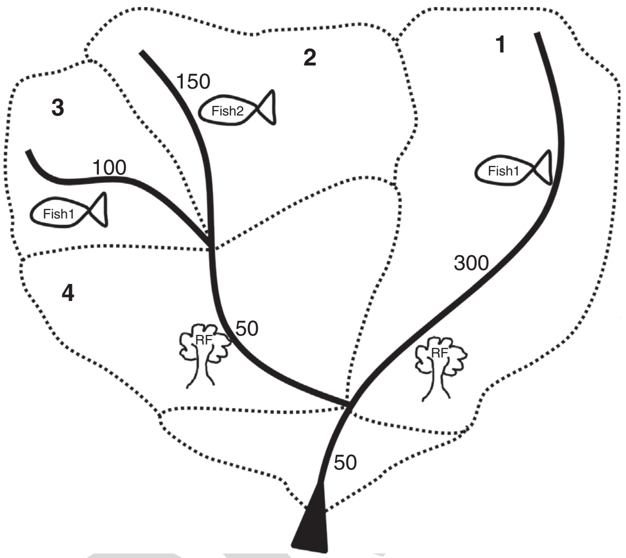

Fig. 4. A simple diagram of how complementarity benefits could be achieved by simultaneously planning for both biodiversity and flow targets. Values show the mean annual runoff per subcatchment. Selecting Subcatchments 2–4 would achieve a desired mean annual runoff of 300 million m^3 for the estuary and represent the most conservation targets – Fish1, Fish2, Riparian forest (RF).

Such integration will require developing new tools that combine the models of environmental flow assessment and complementarity-based conservation planning tools, e.g. Marxan (Possingham *et al.* 2000). A highly simplified schematic shows the potential complementarity benefits (Fig. 4). One of many scenarios may require a desired condition for the estuary, with a mean annual runoff (MAR) flow target of 300 million m^3 per year. In addition, biodiversity representation may require setting targets to capture at least one population of fish1 and fish2, and a stand of riparian forest. Choosing Subcatchment 1 for conservation would provide the desired MAR flow target for the estuary, but would fail to conserve the required suite of biodiversity. Choosing Subcatchments 2–4 would meet the desired MAR flow target and all the conservation targets, as long as there are no large-scale migration dependencies among the fish populations. Catchments 2–4 can also support water resources development in Catchment 1 by providing the natural flooding and sediment pulses to the estuary. In reality, the solutions are far more complex and data-intensive, involving many more biodiversity features and flow-analysis nodes, and increasing levels of uncertainty owing to lack of data. New optimisation tools need to be developed to support the array of data inputs (e.g. expert opinion, surrogate measures of biodiversity, simulations of flow-regime metrics and biodiversity pattern, and flow and conservation targets). These tools should build on the frameworks and decision-support tools developed in environmental flow assessment and conservation planning, which cope with data limitations and uncertainty.

Deriving an overall ecological condition, relative to natural, provides a common currency between environmental flow assessments and freshwater conservation planning, and will be central in the development of tools that combine environmental flow assessment and conservation planning approaches. Recent tools in freshwater conservation planning are heading in this direction, with the development of tools that quantify conservation contribution of any river section (Turak *et al.* 2010) or wetland (Ausseil *et al.* 2010) relative to its ecological condition, and the recognition of multiple-use zones (Abell *et al.* 2007) linked to ecological condition. By quantifying the link with ecological conditions and conservation targets, and the social and economic cost implications of maintaining different ecological conditions, each planning unit can be assigned an ecological condition that achieves conservation and flow targets, while minimising social and economic costs. This approach recognises that altered rivers contribute to conservation targets, albeit to a lesser extent than natural rivers. The conservation planning tool, Marxan with Zones (Watts *et al.* 2009), recently applied in and marine settings (Klein *et al.* 2009*b*), holds potential in this regard.

Linking ecological condition to upstream/downstream cost implications aligns freshwater conservation planning with environmental flow assessments. It also aligns these tools with existing river-health monitoring (Wright *et al.* 1993; Davies 2000; Lazorchak *et al.* 2000; River Health Programme 2006), where monitoring ecological condition can be used to assess the outcomes of environmental flows and conservation planning.

Conclusions

Research and practice in conservation has tended to focus on terrestrial biodiversity, whereas water resources management has tended to have a more utilitarian focus. Given the dire state of freshwater ecosystems and associated biodiversity (Dudgeon *et al.* 2006), it is high time to elevate freshwater biodiversity concerns on the agendas of both these sectors.

There has been a recent steady increase in the development of freshwater conservation planning and environmental flow assessment as catchment-wide planning tools for water resources planning. Integrating these two fields of science is mutually beneficial. Incorporating the outputs of conservation planning into environmental flow assessments could enhance the ecological-assessment component of environmental flow assessments, representing freshwater biodiversity in the catchment more systematically and efficiently. Similarly, freshwater conservation planning would benefit from the value of environmental flow assessment in measuring persistence of ecosystems and species under different flow regimes. Assessment of the social and economic consequences of flow scenarios, within an upstream–downstream dependency framework, would also benefit freshwater conservation planning. Scenario-based environmental flow assessments can also be adapted to show the consequences of a range of different development scenarios on achievement of freshwater conservation targets. Predicted flow responses with climate change can also be assessed for different development options.

If data are at comparable levels of resolution, environmental flow assessment and freshwater conservation planning may be integrated immediately by developing one or more strategic and systematic conservation scenarios by using freshwater conservation planning principles. The effects of different development scenarios on achievement of conservation targets in the catchment may also be assessed. Integration in the medium- to long-term should focus on developing new tools that combine environmental flow-assessment tools with complementarity-based conservation planning tools. Ecological condition will be central in the development of new tools, providing a common currency for assessment, planning, management and monitoring.

Integration of freshwater conservation planning and environmental flow assessment alone will not achieve conservation and sustainable development goals. The exercise must be embedded within integrated water resources management, supported by formal governance and adaptive management processes that complement and strengthen an underpinning philosophy of multi-stakeholder learning and cooperation.

Acknowledgements

We thank the Skukuza group of freshwater scientists who participated in a meeting in the Murray–Darling River Basin to share knowledge and experience on environmental flows, freshwater ecosystem conservation and climate change. Funding for this work was provided by the CSIR, South Africa. We extend special thanks to the guest editor, Richard Kingsford, and two anonymous referees for considerable guidance in the reviewing of this manuscript.

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Manuscript received 26 December 2009, accepted 17 September 2010

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