Title:

A comparative analysis of the Yellow River, China and the Murray-Darling Basin, Australia: an ecosystem services valuation approach to address ecological restoration of rivers.

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Glossary of Acronyms

ABS	Australian Bureau of Statistics
ACEDP	Australia China Environmental Development Program
AusAID	Australian Agency for International Development
BBN	Bayesian Belief Networks
CCICED	China Council for International Cooperation on Environment and Development
CMA	Catchment Management Authority (Australia)
CPC	Communist Party of China
CSIRO	Commonwealth Scientific and Industrial Research Organisation (Australia)
EC	Eco-compensation
EF	Ecosystem Functions
ERC	Ecosystem Reporting Categories (Australia)
ERP	Ecological Restoration Program
EPBC Act	Environmental Protection Biodiversity and Conservation Act 1999 (Australia)
ES	Ecosystem Services
ICEAIMM	Information, Conceptualisation Eco-linking Accounting Investment Management Monitoring framework
IWC	International Water Centre (Australia)
GDP	Gross Domestic Product
GIS	Geographic Information Systems
GL	Giga Litre
GTGP	Green to Grain Program (China)
HRS	Household Regulation System (China)
LAI	Leaf area index
MDBA	Murray-Darling Basin Authority (Australia)
NESS	National Ecosystem Services Strategy (Australia)
NFCP	National Forest Conservation Program (China)
NSIP	National Sustainability Indicators Program (Australia)
NWC	National Water Commission (Australia)
SLCP	Sloping Land Conversion Program (China)
PES	Payment for Ecosystem Services
PRA	Probabilistic Risk Assessment
SDL	Sustainable Diversion Limits
SEEAW	System of Environmental Economic Accounting for Water
SENCE	Socio-Economic Natural Complex Ecosystem (China)
SES	Social-Ecological Systems
SEQ ES	South East Queensland Ecosystem Services approach (Australia)
UNEP	United Nations Environment Program
UNU IHDP	United Nations University International Human Dimensions Programme
YRCC	Yellow River Conservancy Commission (China)

Executive Summary

Freshwater ecosystems have deteriorated at an alarming rate around the world in the last few decades. Many of the strategies being proposed at international forums to halt this decline rely on the concept of ecosystem services (ES). This concept proposes to integrate ecology and economy by quantifying and valuing ecosystems as natural capital, to encourage a more sustainable use and rational basis for trade-offs between the various ES humans derive. The concept has been proposed in an international context, but applying it at national, regional and local scale presents significant challenges, which require an integrated water management approach.

In the first part of this desktop study, a literature review was undertaken to define the problem and analyse the usefulness of ES as a theoretical framework. This is followed by analysing case studies from China and Australia to investigate the level of uptake, the way the concept is implemented or applied, and the potential for knowledge exchange on the topic.

The key challenges of water management in China and the Yellow River Basin were explored by attending the 5th International Yellow River Forum, which provided a broad overview of contemporary topics presented by a wide range of researchers. These challenges were linked to the concept of ES in China, and further research publications were identified that provided relevant analysis and recommendations on the topic. Similarly, key water management challenges were investigated and related to the work undertaken on ES in Australia and the Murray-Darling Basin.

The concept of ES can be viewed as a tool to assist with the protection of freshwater ecosystems, by:

- quantifying the extent and nature of the ES that are provided
- providing a framework where these services can be compared with other potential demands on the resource, some of which will compromise sustainability of the ecosystem
- value the ES in an economic context, so that trade-offs can be made based on a rational decision
- internalise the costs of degradation, to assist with identifying strategies for freshwater ecosystem protection

Implementing an ES framework requires an integrated water management approach. Valuation of ES will require input from stakeholders, participatory governance, and expert knowledge from ecologists, economists and hydrology modellers to define and quantify the trade-off decisions. In order to prioritise conservation and protection of freshwater ecosystems, ES frameworks are one useful tool, but require complementary measures to make it effective for ecosystem protection.

Based on an analysis of the literature a new framework is proposed. This framework links ES quantification with on-ground river restoration projects, and consists of seven stages: 1. Informing, 2. Conceptualising, 3. Eco-linking, 4. Accounting/Trade-off, 5. Investing, 6. Managing and 7. Monitoring/Evaluation (ICEAIMM).

Examples of how the framework could be applied in the Yellow River Basin, China, and the Murray Darling Basin, Australia are provided. The proposed framework is then assessed for its strengths, weaknesses, opportunities and threats. The framework provides opportunities for Australia to demonstrate how ES can be linked from a global to a local scale, whereas China could promote its payment for ecosystem services (PES) and eco-compensation (EC) schemes. Finally, three key recommendations are proposed:

Recommendation 1:
ES should be linked explicitly to restoration and conservation frameworks, in order to integrate the strategy of 'stock taking' and valuation of ecological capital with strategies to preserve and maintain this capital.

Recommendation 2:
The proposed ICEAIMM framework provides a mechanism to integrate ES with ecological restoration and conservation. It is recommended to explore the use of this framework to implement Recommendation 1.

Recommendation 3:
Dialogue should be maintained between Australia and China about water resource management, to foster opportunities for knowledge exchange, and to work jointly on integrating national ES projects under the international framework as proposed by the Millennium Ecosystem Assessment.

There are some things in the world we can't change – gravity, entropy, the speed of light, and our biological nature that requires clean air, clean water, clean soil, clean energy and biodiversity for our health and well-being. Protecting the biosphere should be our highest priority or else we sicken and die. Other things, like capitalism, free enterprise, the economy, currency, the market, are not forces of nature, we invented them. They are not immutable and we can change them. It makes no sense to elevate economics above the biosphere.

David Suzuki.

Table of Content

1. Objective of the paper

In this paper the ecosystem services (ES) concept is explored as a tool to assist with protecting our valuable freshwater ecosystems. The ES concept was proposed as part of the Millennium Ecosystem Assessment (2005) to assist with the protection of biodiversity for human well-being at a global scale. Implementing an ES framework on the ground requires an integrated water management approach. Case studies in China and in Australia are explored to examine:

- the level of uptake of the ES concept
- country and Basin specific application of the concept;
- a proposed integrative framework for the use of the ES concept with examples for application in Yellow River, China and Murray-Darling Basin, Australia;
- an evaluation of the strengths, weaknesses, opportunities and threats of the proposed framework
- the potential to link on-ground implementation back to national and global scale frameworks.

2. Method

This work was undertaken as a desktop study. The usefulness of ES as a framework to assist with the protection of ecological capital was analysed by way of a literature review. Attending the 5th International Yellow River Forum enabled me to explore the key challenges of water management in China and in the Yellow River Basin. I linked these challenges to the concept of ES, and identified further research publications that provided relevant analysis and recommendations on the topic. Similarly, I investigated key water management challenges for Australia and the Murray-Darling Basin, and related this to the work undertaken on ES in Australia.

Implementation of an ES framework on the ground requires an integrated water management approach, for which a new integrative adaptive management framework is proposed. This framework combines ES with ecological restoration/conservation programs. The framework is applied to the Yellow River, China, and the Murray-Darling Basin, Australia, to illustrate how this can be used. Finally I discuss opportunities for integration at the global scale and the contributions China and Australia can make.

3. Introduction

3a. Problem description

Freshwater ecosystems are in rapid decline worldwide, due to the pressures from excessive demand on this natural resource for multiple human uses. The living planet index, an indicator of the state of the world's biodiversity, indicates that the extinction rates since 1970 have been worst for freshwater species (50%), compared to terrestrial (30%) or marine species (30%) (Millennium Ecosystem Assessment 2005). Impacts are greatest in areas with high population densities, where water security is often a serious issue as well. However, securing water for consumption often aggravates impacts on biodiversity and ecosystem functions through infrastructure development (Vörösmarty et al. 2010).

Global population growth has more than doubled in the last 40 years which, together with the rise in living standard, has accelerated pressure resulting in a range of adverse impacts, including: loss of biodiversity, declining condition of vegetation, salinisation, algal blooms, deteriorating water quality, invasive pest species, excessive sedimentation and erosion, loss of floodplain and wetland flows and habitat loss in general (De Groot 2012; Worldwatch Institute 2013; Millennium Ecosystem Assessment 2005).

With this realisation comes the awareness that ecosystems and their functions provide many beneficial ES to humans, and that there is a need to manage ecosystems in a sustainable way. Sustainable management of natural resources (and freshwater systems in particular) requires knowledge on:

- What ecological capital (the extent of a particular ecosystem and its biodiversity) is present, and how is its quantity changing over time?
- What are the pressures on this ecological capital with regard to required services?
- What is its current condition, and how is this changing over time, as a result of direct and indirect human impacts?

The concept of ES, provided by ecosystems and their functions, enables us to develop a typology to classify, take stock, assess and potentially quantify the value of this ecological capital for present and future generations (De Groot et al. 2002). ES provides the link between the complex functioning of ecosystems (which provide the supply of services) with human uses, demand and economic dependencies. Human demand is embedded in an economic system that has its own dynamic complexity.

Ecological economics is a relatively new discipline which attempts to value ecology, but does so by exploring a deeper understanding of the way in which economic activity depends on biophysical processes. This is done by integrated ecological-economic modelling and assessment (Costanza 2008).

The concern for the degradation and sometimes failure of river systems to cope with the cumulative impacts of human water uses has prompted many governments around the world to monitor for ecosystem health as a first step in conservation and sustainable management. However, monitoring efforts are disparate, lack coherence and consistency, and are inadequate or non-existent when it comes to dealing with cross-border issues (Lindenmayer et al. 2012; Lindenmayer & Likens, 2009). As a consequence, it has been difficult to measure long term trends and hence the effectiveness of conservation programmes.

Concepts of ecosystem health have evolved from the Arcadian naturalness model, where nature is defined as 'pristine' and unimpaired from human actions (Schaeffer et al. 1988; Anderson 1991; Angermeier & Karr 1994; Wicklum & Davies 1995) over systems theory – dynamic systems are irreducible fundamental units – (Wolfgram 1984), succession-to-climax models (Odum 1969), to non-linear dynamics, multiple equilibria and resilience (Holling 1973), the diversity-stability model – greater diversity equals greater stability (Batabyl 1998; Ferreira & Towns 2001), the emergy, exergy and ascendancy models – measuring energy to build and operate the system, and network complexity (Jorgensen 1997; Ulanowicz 1980; Ulanowicz 1986; Wulff et al. 1989) and finally the order, chaos and complexity – complexity emerges out of chaos (Kay 1984; Kay 1991) and the adaptive cycle model – cycling stages between *K* and *r* strategies (Holling & Gunderson 2002). Concepts dealing with stressors include the stress induced community tolerance concept and the multiple stressor index (Tockner et al. 2010).

Most of these conceptual models describe ecosystems that are characterised by all or parts of the following:

- The dynamics of the system are non-linear.
- Ecosystem functions are responsible for energy and material cycling through the system.
- States of equilibrium exist, but also states of non-equilibrium or even tipping points to other equilibria.
- Ecosystems constantly change; resilience or 'elasticity' is the degree of flexibility a system has to return to equilibrium.
- Systems are made up of networks of interacting populations.
- Controversy exists around whether greater biodiversity, which has intrinsic value in its own right, means greater resilience; this hypothesis remains unresolved.
- Ecosystems are open (nested) systems that can exist in states of non-equilibrium.
- Ecosystems are self-organising systems, with emerging properties resulting from interconnectedness.
- Ecosystems go through adaptive cycles which require transitions from chaos to order and vice versa in stages of:
    - exploitation (*r* or production phase);
    - conservation (*K* or accumulation phase);
    - release (*Ω* or invention phase); and
    - reorganisation (*α* or re-assortment phase).
- Human induced stressors can be identified and used to classify ecosystems for conservation management.

Given that ecological systems are dynamic, self organising systems that can alternate between different states, categorical statements about an ecologically "good" state for an ecosystem cannot be deduced from "objective" scientific arguments (Kuhn 1962). Therefore, ecological health cannot be defined by ecological criteria and is essentially a normative concept. Some propose that it can be determined by economic means alone (Hearnshaw et al. 2010), but this ignores the fact that our economic frameworks are ill equipped to account for complex, non-linear systems which may respond to human pressures in unpredictable ways. It also ignores the fact that condition and extent of ecosystems are inextricably linked, given that ecosystems are self-organising systems for which a critical 'mass' of biodiversity is required.

ES can be defined as services providing benefits for humans in a direct or indirect way from biodiversity and ecosystem functions; these can be categorised as provisioning, regulating, cultural and supporting services (Millennium Ecosystem Assessment 2005). Provisioning services typically include direct natural resources 'goods', such as wood, water, fibre, fuel, fish and so on. In developing countries, subsistence economies typically will be highly dependent on these services (i.e. low income families may be dependent on it for their survival). Regulating services ensure that favourable conditions continue to exist (for example clean water, stable climate, control of disease vectors and so on). Cultural services include spiritual, aesthetic, recreational and education, whereas supporting services are those that maintain ecosystem provisions (soil formation, primary and secondary production, to name a few); the latter can be seen as ecosystem functions underlying all other services.

Mace et al. (2012) define the progression as ecosystem processes (e.g. soil formation, nutrient cycling, primary production, biomass and pollination) resulting in ES (e.g. clean water provision, crops, trees, water regulation), which translate into goods (e.g. drinking water, cereals, meat, timber, wild bird species, flood protection), which are then valued depending on added inputs and on context. Very few studies exist that link changes in biodiversity with changes in ecosystem functioning and changes in human well-being.

The relation between biodiversity and ES is complex. On the one hand biodiversity can be regarded as equalling ES (ecosystem services perspective), on the other, biodiversity can be regarded as one among many ES (biodiversity has intrinsic value – conservation perspective). Neither perspective is consistent with ecological science (Mace et al. 2012): the former ignores the intrinsic value of biodiversity, and the many processes other than those providing ES; the latter ignores the role of biodiversity in underpinning ES. The authors conclude that effective ecosystem management will require identifying and understanding the roles relevant for both optimisation of ES delivery and for the conservation of species, habitats and landscapes. They strongly advocate new approaches underpinned by ecological science, in order to understand and quantify trade-offs and synergies in these coupled human-environment systems (Mace et al. 2012). In a practical sense, the premise to conserve biodiversity to maintain ecosystem integrity is a valid working hypothesis that is adopted in most ES frameworks. When classifying and comparing ecosystems and their associated ES, it is important to ensure classification schemes are compatible in the way biodiversity is defined.

3b. Ecosystem valuation

From an economic perspective, ES to date have been considered externalities based on the assumption that their supply was limitless. The current rate of modification of those systems, rising pressures for water demand, and the resulting drying up of rivers and collapse of lake systems illustrates that this assumption is flawed and needs to be revisited. In order to make decisions on their sustainable management, ecosystems and biodiversity need to be considered as ecological capital; this capital generates ES, many of which cannot be substituted by human-made systems or production, while others are crucial to our survival as a species.

However, internalising ecosystems and ES risks making them victims of the limitations of our current economic growth model, where marginal benefits and losses are determining the extent to which ecosystem capital is compromised on a global scale. This growth model is unlikely to be sustainable in the long term, given that we are depleting our ecological capital at an unprecedented rate, with extinction rates 1000 times that of the fossil record (Millennium Ecosystem Assessment 2005). Moreover, the intrinsic value of ecosystems cannot be valued in conventional economic terms.

Defining ecological systems in terms of ES is one stream of thinking that has attempted to quantify their value, starting by mapping and classifying various major ecosystems, in order to quantify their extent. However, standardising classification systems at a national and international level remains a high priority (Jian 2011). The ES research review in China by Zhang et al. (2010) classifies ecosystem types into seven major categories. Out of the 230 evaluations identified, most studies are on forests ecosystems (104), followed by regional ecosystems (48), wetlands (31), grassland (17), rivers (13), farmland (11) and marine ecosystems (6). The valuation of the 'products' is done by different methods, which makes their comparison dubious.

The ES concept is also difficult to reconcile with the current economic valuation techniques, which are reductionist in nature and fail to capture the value in terms of internal systems integrity. Yung En Chee (2004) has appraised the use of traditional economic valuation techniques to value ES (production value of ES, replacement/restoration value, travel cost method, hedonic pricing and contingent valuation or hypothetical market value), and concluded that a range of other techniques are required to address their shortcomings. Instead of top-down evaluation, the author recommends social learning, citizen science and participatory approaches, modelling and probabilistic risk assessment (PRA), and Bayesian Belief Networks (BBN) among others, used in the context of real world problems requiring multiple criteria and constraints. Everard et al. (2009) apply the concept of value chains to specific ES. A value chain provided by a natural ecosystem can be modified by intervention management, generated from collective visioning of a clearly articulated goal. This goal can take into account multiple interests and criteria, and is then used to 'back-cast' the step changes required in an adaptive management process, which reflects a desired value chain and provides desired ES. The planning process will need to be adaptive, as a linear command and control approach reduces variability and resilience, and hence creates imbalance (Briggs 2003). In addition, periodic revisions of the management plans against objectives will need to be integral to the adaptive management cycle, given the sobering lessons learned from past experiences where land care strategies based on causal attribution were proven not to work (Lefroy et al. 2012). Reasons for failure included poorly defined baseline conditions, interventions undertaken at the wrong scale and in the wrong locations, and multiple interacting drivers instead of singular cause and effect mechanisms. The study examined the effectiveness of three types of information (land managers experience, expert opinion and quantitative data) at three spatial scales (landscape pattern, property, and site scale). It concluded that qualitative data and social research 'trumped' in terms of accuracy despite ecologists' reservations about relying on this type of information (Lefroy et al. 2012).

The ES approach consists of measuring, mapping and valuing ES (Primmer & Furman 2012). Using this approach for management strategies depends on this quantification process, which faces considerable challenges:

- Collecting information on extent and ecosystem functions will always remain incomplete and provide a picture of past up to present condition. This information may not be helpful in determining future ecosystem responses to human pressures, given that ecosystems are dynamic systems that can respond in non-linear ways. Human pressures may significantly reduce our capacity to manage for sustainable outcomes or conservation.
- Spatial heterogeneity means that ES are rarely accessible in a homogenous way, and are often not proportional to ecosystem size (De Groot et al. 2012). In fact ecotones (borders of different biomes) often provide more diversity and hence services than the ecosystem 'interiors'. These areas are often highly valued and consequently modified by humans (floodplains are a good example, where the nutrient rich depositions provide fertile soils for agriculture). Costanza et al. (1997) state that it is important to map the precise delineations of ecosystem boundaries. However, the spatial delineation of ecosystems will depend on the resolution and may overlook the importance of ecotones which are often at greater risk of human modification (for example riparian vegetation is frequently at greater threat of degradation from cattle grazing and weed infestations than either terrestrial or aquatic vegetation).
- Valuation of ES faces challenges regarding a wide range of valuation methods, difficulties in isolating values (which may result in double accounting), values that are time and location specific (and do not include unknown future values), beneficiaries at different scales, selection bias of value estimates, interactions between service use and influence of management (De Groot et al. 2012) and ecological knowledge gaps on species and the habitat niches they occupy in an ecosystem. Braat & De Groot (2012) suggest that in order to better understand quantitative relationships among aspects of biodiversity, ecosystem components and processes, functions and services, two potential indicators are required:
    - State indicators: describing which ecosystem process or component is providing the service and its quantity (e.g. biomass or leaf area index LAI);
    - Performance indicators: describing how much of the service can be used in a sustainable way.

While the development of the ES concept is still in its infancy in many parts of the world (due to the challenges listed above), operationalising them faces its own set of challenges. Primmer & Furman (2012) discuss how measuring, mapping and valuing could integrate sector specific knowledge in the context of operationalising ES in Finland. Key findings from their discussions are:

- Measuring all or a very broad range of ES is unrealistic. Moreover, in many cases additional information does not directly lead to use of this additional knowledge.
- Therefore, operationalising of ES valuation techniques requires more work in governance settings, rather than distinct research activities; it should build upon existing knowledge systems and governance arrangements but aim at communications across ecosystem and sector boundaries.
- Knowledge production should be organised to service collectively identified minimum knowledge needs that would serve many sectors and many actors governing many ecosystems.
- Measuring and mapping should be supplemented with more detailed local level analysis of ES.

The use of the ES concept to protect and restore ecological 'capital' is only one step in a very complex process, concerned with quantification. Other, but equally important dimensions are cost effectiveness of rehabilitation planning and implementation, systemic and long term condition monitoring, protection strategies, the international and national governance context, private/public ownership of ecological assets, and stakeholder participation. Hermoso et al. (2012) discuss the low success rate of river rehabilitation, and propose a systematic decision support system for river rehabilitation that offers integrated solutions to stakeholders using optimisation methods that show the best trade-off between a set of competing objectives. This process based method should be used at the catchment scale and be part of an adaptive management cycle. Further, we need to be conscious of the characteristics of complex systems, which are open-ended, difficult to define, and have solutions that are dependent on how the problem is formulated (Rittel & Webber 1973).

Internalising ES into the current economic framework will require active policy development and creation of incentive schemes. Further, there will be a need to link ES quantification internationally (similar to the schemes addressed to curb carbon emissions in response to climate change), so that we can assess ES that are operating on a global scale.

It is clear from the problem description as outlined above that singular disciplinary perspectives are not sufficient to manage freshwater ecosystems in a sustainable way. Integrated Water Resource Management will need input from ecological science, economics, social learning, hydrology, engineering, stakeholder engagement, good governance and policy setting to reconcile human demands and trade-offs for the various uses of water and water related ES, at a variety of spatial and temporal scales.

4. Ecosystem services application in China

4a. Water resource management issues in China

The China Council for International Cooperation on Environment and Development recommended the need for Integrated River Basin Management as a way to solve ecological degradation issues in the nation (CCICED 2004).

The urgency of the need to address water management issues including ecological protection and conservation was recently recognised by the Chinese Government, which released the No.1 Policy document in 2011 entitled:
"Decision of the Central Committee of CPC and the State Council on Accelerating the Water Conservancy Reform and Development" (CPC Central Committee 2011).

This strategic document is important as it is the first time that the annual policy released by the Chinese government addresses water reform as a priority objective, with clearly defined targets to be achieved over the next five to ten years. It identifies population pressures, scarcity of water resources, uneven spatial and temporal distribution, water pollution and frequent flooding as bottlenecks for sustainable development, and lists water conservancy as a national priority.

The document lists a wide range of measures to be undertaken as part of water governance reform (30 articles), but its wide scope makes it difficult to ascertain which of those will take priority in terms of decision making, governance support and resource allocation. At the 5th International Yellow River Forum in Zhengzhou, (September 2012) some priorities were presented, known as the 'three red lines': (a) strengthen water resources development and utilisation and control the total amount of water use, (b) promote water use efficiency, and (c) control of water pollution and effluent discharge to within carrying capacity (Guoying 2012).

Liu & Speed (2009) discussed the legal and institutional frameworks that exist for dealing with the three principal challenges in water resource management in China: water scarcity (b), pollution (c) and flood control (a). Competing water demand across agriculture, urban and industrial water uses and the inequitable distribution of water across many regions in China requires targeted water allocation plans, which are currently embedded within legal frameworks at basin/regional level, abstractor level and within public water supply systems.

Shen & Speed (2009) identified problems with integration and consistency across the three allocation levels in the context of determining environmental flows. Ecological flow requirements are poorly understood in China and where allocations are made they are usually set as a fixed percentage (or volume) of the total water resource. The aspirational targets under water resource allocation plans mean that they are often not fully implemented. Regulatory instruments should therefore replace aspirational targets and an inventory of key ecological assets and their flow requirements should be determined, together with a quantification of consumptive water use requirements (volumes, reliability and options for supply) (Shen & Speed 2009).

The concept of ES in water resource management is closely linked with the idea of environmental flow regimes required to support ecological structures and functions, which in turn will provide services including clean water for consumption. The latter is the most critical service required by humans, and many debates therefore quickly focus on the amount of water that can be allocated for consumptive use, and tend to overlook the importance of non-consumptive uses which can nevertheless constitute an important part of the economy (e.g. tourism).

In China, water demand has increased by 32% between 1980 and 2007, and has produced competing uses across inequitable spatial scales. Agriculture is still the main water user (65%) followed by industry (23%) and domestic use (12%) (Liu & Speed, 2009). Increased water demand, which has reduced assimilative capacity, and the doubling of wastewater discharge during that period has resulted in severe degradation of the water quality, with more than half the cities in China no longer fully complying with drinking water standards (Zhang et al. 2010). In addition, water shortages are frequently common, and projects are being planned for South-North water diversions, including from the Yangtze River to the Yellow River (Dong et al. 2011).

4b. The concept of ecosystem services in China

The concept of ES, as defined in an international context, gets translated in China in a different way than in other parts of the world. Since people have inhabited China for many thousands of years, the concept of viewing the environment as separate from humans (which was developed in an international context) is difficult to embed in Chinese thinking about natural resource management. In China, the definition of ecological water requirements would include "not only maintenance of biodiversity and natural ecosystem function, but also maintenance of landscape as a place for human habitation and livelihood" (Giordano et al. 2004).

In 1996, the State Council of the People's Republic of China recognised the potential for using economic and ecological compensation, as well as payments for environmental services to protect natural resources and restore ecological functions. Eco-compensation (EC) is hence a well established concept in China, and large sums of money have been spent with China leading the way in the use of EC schemes. The Natural Forest Conservation Program (NFCP) and the Grain to Green Program (GTGP) are the most well known; they covered 97% of China's counties and are the largest payment for ES programs in China and in the world in terms of scale, payment, and duration (Liu et al. 2008). However, there are also successful eco-compensation projects related to freshwater ecosystems, such as the Huangshan industrial restriction initiative (China Green News, 2013).

EC in China is broadly defined as an institutional arrangement that regulates the distribution of ecological and economic interest among all stakeholders. In reality, it is mainly used for compensating stakeholders for economic opportunities lost as a result of government policy decisions for the management of natural resources, and is far less common for payment for ecological services (PES) (White 2011).

The indicator categories developed for forest ecosystem assessments are water conservation, soil conservation, carbon fixation and oxygen release, nutrient accumulation, atmospheric environmental purification, action of forest against natural calamities, biodiversity conservation and forest recreation. Many of these indicator categories are also relevant for freshwater ecosystems, where they could be applied to measure functional and structural integrity of the latter. There is certainly potential to develop standardised classification systems and indicators across the various ecosystems in China, as recommended by Zhang et al. (2010), who also advocate that in future assessments regional dynamic models need to be constructed to reflect the spatial heterogeneity of ecosystems, and a comprehensive accounting system for valuation and environmental decision-making tailored to the Chinese context.

Yin and Zhao (2012) explore the concept of 'payment for ecosystem services' (PES). They advocate that in order for these investment schemes to be effective, they need to be integrated with ecological restoration programs (ERP). The complex interaction between ecosystems and socio-economic systems requires an integrated approach, along a diagnostic action framework developed by Ostrom (2007) for dynamic social-ecological systems (SES). The framework identifies four components that influence a particular action situation: (i) the resource system (ecosystem), (ii) the resources services and units (ES), (iii) the governance system and (iv) the actors. Importantly each of those components will influence an action with feedback loops that can make outcomes differ from intended purposes. The framework is nested within a larger spatial and temporal socio-economic as well as ecological context.

Yin & Zhao (2012) apply this framework analysis to the Sloping Land Conversion Program (SLCP), better known as 'Grain for Green', one of several large ERPs launched in China, which subsidised farmers to undertake restoration activities in 25 provinces in the last decade. Several unintended effects of the program were identified, which could have been avoided by using a systems approach for research and planning:

- Large scale implementation without pilot projects;
- Poor consultation with actors (farmers);
- Not integrated with farming management practices;
- Poor administrative budgeting;
- Poor selection of indicators to measure restoration outcomes;
- Monoculture conversion with poor effectiveness results;
- Accelerated erosion on abandoned land plots.

The authors conclude that successful restoration programs require an integrative assessment across multiple spatial scales which embraces both environmental (ecosystem productivity and stability) and socio-economic (cost effectiveness, labour transfer and livelihood enhancement) changes in an interdisciplinary fashion. Dong et al. (2011) identified poor criteria for payment as a deficiency that needs to be resolved for most PES programs; they developed improved criteria, identified water related ecosystem service users as clearly identifiable beneficiaries and developed a watershed criteria model for the Middle Route Project of South-to-North Water Diversion.

4c. The Australia China Environmental Development Program

The Australia China Environmental Development Program (ACEDP) that ran from 2007 to 2012 included a large eco-compensation project in partnership with the Chinese Ministry of Environmental Protection, the Ministry of Water Resources and the Australian National University. Its aim was to assist acceleration of nationwide EC mechanisms by providing EC policy options and mechanisms. The project undertook an analysis and review of environmental, natural resource and agricultural policy and regulations, public administration principles, current EC and PES programs. It ran two EC and PES case studies in China in trans-boundary river systems, and two in Australia (White 2011). Recommendations emerged from the project in five key areas:

- Design principles should be based on public policy and regulation theory and principles of public administration, have clear and measurable EC policy objectives, the implementation mechanism should be reliable, predictable, open and transparent, accountable, efficient and effective, and include the principle of subsidiarity, which states that decisions need to be made at the lowest possible level.
- Priorities should focus on trans-boundary tributaries and lakes, national key ecological areas and major inter-basin transfer projects. The initial focus should be on ES related to water yield, quality, flood detention and erosion control, in areas used for large city domestic water supplies.
- Financing EC schemes should be based on self financing. Mechanisms could include a national EC fund financed through variable rate water abstraction charges (increased over 10-20 years), and incentives using tax concessions for industries, regions and individuals adopting water saving and water quality protection technologies.
- Governance and management recommendations include establishment of integrated cross sectoral and jurisdictional organisations for trans-boundary catchments, key ecological sites and inter-basin water transfers, and a system of water entitlements at province, municipal and local level.
- Assessment frameworks need to be created as a matter of urgency, to ensure public funds are well spent, ecosystem functions are being protected and improved, and desired environmental and social outcomes are achieved.

A model framework and an assessment tool were developed, to assess the suitability of selected projects, targeted ES are being delivered, equality and socially coordinated growth is being achieved, and ES schemes are efficient and effective.

Overall, the major findings from this project were that the case studies, run as cooperative learning exercises, were an effective way for capacity and partnership building.

Other related projects from the AusAID funded ACEDP program were environmental flows and river health projects (Speed et al. 2011). They are relevant because PES needs to be integrated with ecological restoration and monitoring if the aim is to protect, restore and manage ecological capital in a sustainable way. The findings from the ACEDP synthesis report, which did an evaluation of all 23 projects that were part of the program, was that international collaboration over the five years offered complementary approaches with significant potential to build on the engagement, though new funding mechanisms would need to be identified (AusAID 2012). The ACEDP program has contributed to environmental and water reform, not only technically but also by advancing concepts of adaptive management, subsidiarity principle, transparency and genuine public participation in China.

4d. Ecosystem Services in the Yellow River.

The Yellow River features as an important natural resource as well as a cultural and historic icon in China. It is considered the river around which the Chinese nation took shape, and has been named the 'cradle of the Chinese civilisation', as well as 'China's sorrow', due to the massive floods that have caused so much hardship in Chinese history. The Yellow River is the second longest river in China (>5400km long) and its catchment contains about 9 per cent of the population of the nation and 17 per cent of its agricultural area (Giordano et al. 2004). The river can be divided into three main reaches. The upper reach consists of two parts: the river's origin in the Bayenkala mountains to Lanzhou contributes 56% of the total run-off (Song 2011), followed by the part that runs north into the Ningxia/Inner Mongolian plains and the Gobi Desert, where evaporation rises to levels several times that of precipitation, and the river becomes a net consumer of runoff (World Bank 1993). The middle reach runs through the Loess plateau, responsible for the majority of its huge sediment load. The lower reach, which has a suspended river channel relative to the surrounding floodplain, runs through the most populated area of the Basin (Song 2011).

The Yellow River has undergone a rapid transition since the 1970s, as a result of accelerated economic development, with incremental and unintended change requiring an integrated management approach (Webber et al. 2008). The key management challenges resulting from increasing demand are related to ES people have been relying on for many hundreds of years:

- Water provision for drinking water purposes and for irrigation along its main course in a equitable way
- Discharge of waste waters (mainly household and agricultural wastewaters in the past, but increasingly also industrial wastes)
- Environmental flows for the wetlands in the lower delta of the river
- Mitigation of flooding through erosion control and management of sediment deposition in the lower reach

The arid and highly variable climate has been magnified in the last 40 years, and has resulted in no-flow days in the middle and lower reaches of the river, affecting among others the fragile lake ecosystems in the river delta, continued risk of flooding in the lower reach, and water pollution, which is accelerating rapidly.

Webber et al. (2008) identify four key drivers of this change:

1. Household responsibility system (HRS), which created the possibility for farmers to increase their income by intensifying agricultural production. This has not only increased competition for irrigation water but also undesirable features, such as fertilizer run-off, among other things.
2. Urbanisation, which has reduced allocations for agriculture, decreased aquifers around the city, and shifted demand to less efficient crops as a result of higher income.
3. Industrialisation, which has increased water demand and significantly contributed to water pollution.
4. Changing geography of agriculture, resulting from industrialisation in the South, with agriculture shifting towards the drier parts of China and hence increasing irrigation demand.

The authors conclude that these changes are taking part in a national context (thus are driven by external changes relative to the Yellow River Basin), and have had a negative impact on management cost (water transfer solutions) and ecological cost.

The Sloping Land Conversion Program (SLCP) was a Payment for Ecological Services restoration program that attempted to improve soil stabilisation and reduce erosion through providing financial incentives for converting cropland on steep slopes to grassland and forest (Zhang et al. 2010). A significant reduction in suspended sediment load would be a long term management aspiration to reduce the risk of flooding in the lower Yellow River, which occurs as a result of the combination of a suspended stream bed and narrowing stream channel. However, relevant indicators used under the SLCP program such as water and soil conservation nutrient accumulation, would need to be integrated in a water ES framework by measuring the same parameters in the river, and setting targets that mirror what is required on land. Water quality targets will also need to be integrated with quantitative targets.

A water allocation regime for the Yellow river will need to reconcile several competing water uses, including irrigation, urban and industrial water supply, and environmental flows. Shen and Speed (2009) discuss the many water allocation management challenges in China and in the Yellow River in detail. Among other things, they identified the need for including environmental flows into allocation plans, based on in-stream ecological requirements, which can be integrated through water resource management models and decision support tools. Defining environmental flows based on in-stream ecological requirements is relatively new to China, and would complement the current definition which focuses on flows required for sedimentation flushing (AusAID, 2010).

5. Ecosystem Services application in Australia

5a. Ecosystem services in Australia

In Australia, Pittock et al. (2012) provide some insights in how the ES concept has been applied nationally. Pioneering approaches have been applied in better valuing ES, such as conserving catchments for water supply. In integrated catchment management, a wide range of stakeholders have been engaged for managing natural resources using an ecosystem framework; this has also been extended to regional land use policy, including larger scale developments such as indigenous land and sea management institutions. But using ES frameworks in facilitating dialogue within and between state and federal governments has been much less successful, resulting in a lack of commitments in implementing policy to prevent the ongoing decline of ecosystems at the larger spatial scales. ES have been managed piecemeal (e.g. water markets), creating new externalities between, for example, water and forest mechanisms. Many of the ES that have been obtained in Australia have resulted from converting ecosystems for agricultural production. But the adoption of ill-suited European agricultural practices has resulted in the long term decline of many ES, including salinisation, eutrophication and so on. The authors conclude that at the heart of ecosystem decline is the view of biodiversity competing with socio-economic benefits, resulting in compromises which further degrade ecosystem condition (Pittock et al. 2012).

There is certainly a tendency to simplify important debates on natural resource management issues to key contrasting benefits and required trade-offs, rather than considering a more comprehensive view of all beneficial ES and potential trade-offs. Such a view would nuance the debate considerably and reduce the possibility of the debate being high-jacked by powerful interest groups. The National Water Commission advocates such an approach, in a publication covering the ES best practice approach (National Water Commission 2012a), where it is postulated that the trade-off should be seen as between water consumptive use versus public non-consumptive ES benefits (in addition to the protection of ecosystems). Increasing the consumptive use of water also increases the risk of system failure (ecosystem processes and functions), with concomitant loss of ES. In its publication a benefits table (matrix) is proposed to achieve the trade-off, which uses program logic (a series of interlinked logical steps) to populate the table in a number of sequential steps (Appendix 1).

Whereas this approach provides a more logical approach for looking at ES it is problematic for a number of reasons:

- The non-inclusion of intrinsic value of ecosystems and the concept of trade-offs suggests the dichotomy between humans and ecosystems continues; beneficiaries other than humans (fauna and flora, ecosystems themselves) are not counted in the valuation process.
- Valuation is context specific and cannot be generalised or extrapolated over different spatial scales. In addition, it is costly and time consuming and hence cannot be undertaken in a comprehensive manner. It is therefore unrealistic

to assume that this valuation tool will be adopted as a standardised approach for water planning purposes, nor that it will be applied comprehensively.

- The tool advocates populating the benefits table with input from beneficiaries. This begs the question who decides who are the beneficiaries, and how are these selected for participatory planning? This process is not explained in any detail, and adds further complexity to the planning cycle.
- The protection of ES is not addressed explicitly in the benefits table. It is assumed that reducing consumptive water use will result in better environmental outcomes, but ways to measure this or look at other pressures on these systems are not discussed. Therefore, environmental outcomes are tied to specific objectives assumed to correlate with key ecosystem functions (such as maintaining x% of flow.) rather than broader ecosystem protection or restoration. This narrow definition of ecosystem health needs to be complemented by landscape scale baseline assessments and planning for adaptive management.

Future directions advocated in this work are not less problematic:

- Translating ES concepts into planning practice is recognised as 'cutting edge' work (Daily & Matson 2008). Given the findings from Pittock et al. (2012) on how little the uptake has been in Australia, one is left to wonder if there will be sufficient momentum to make this conceptual approach more influential.
- More information is required on non-commercial and less tangible benefits, resilience or sensitivity of a 'service', and on existing information for a local area. Given that this approach is costly, the question arises what capacity will be available to accomplish these objectives.
- Spatial mapping of processes, services and benefits supplied by aquatic systems is required.
- Dealing with complexities will require making simplifications. This is already occurring by assuming linear responses. We know however that interactions are often non-linear and that tipping points exist (Holling 1973; Holling & Gunderson 2002). The risk of simplified assumptions is that they take on a 'life of their own', and that management actions based on them may not adequately protect the ecosystems that are intended to be safeguarded.
- Decision criteria for reasonable investments should be based on values rather than valuation in the narrow economic sense only. Problems arise because the benefits table only deals with contemporary beneficiaries. The table does not assign values for future uses (which may often be unknown). Therefore the trade-offs will be biased towards current stakeholders (if these can be selected in an equitable way, see point above).
- The other (complementary) approach is resilience thinking (Smith 2012), which may be better suited to deal with future prospects of an ecosystem. Resilience can be defined as the ability to absorb disturbances and reorganise into a better configuration while still retaining fundamental characteristics of a particular ecosystem (Smith 2012).
- Accounting and performance based systems, when linked with water accounts, can potentially combine top-down and bottom up approaches in a

governance mechanism. This leads us to a discussion of environmental accounting.

5b. Environmental accounting in Australia

Environmental accounting in the water sector in Australia is based on the SEEAW framework, as advocated by the Inclusive Wealth Report (UNU-IHDP & UNEP 2012). The United Nations System of Environmental Economic Accounts for Water (SEEAW) is an important step forward in understanding flows (abstraction, consumption and return flows) and stocks (including groundwater resources, lakes and snowpack) which are harder to assess. SEEAW is multi-sectoral, distinguishes between consumptive and non-consumptive uses of water, is hydrologically consistent and can be applied at various spatial scales (UNU-IHDP & UNEP 2012). The framework has been adopted by the Australian Bureau of Statistics (ABS 2012]. The Bureau of Meteorology is also using the framework to collect systematic data on water assets.

Despite SEEAW actively been promoted internationally, Australia seems to be the only country where detailed implementation plans and progress seems to be available. However, even in a sophisticated and highly developed economy such as Australia the framework remains difficult in application. For example, forestry, a key area of water diversion is not included in the water accounting framework. Studies by Zhang et al. (2001) suggest that plantation reforestation is a considerable consumer of runoff, which will reduce the annual runoff in parts of the catchment.

Quantifying stocks accurately is another challenging area. The accounting strategy, which aims to generalise the state of the resource in terms of efficient use, does not account for what is needed to maintain ES, so the framework will need to be extended to account for the environment. In conditions of scarcity and competition, the hydrologic context is critical, and simplistic statements about saving water, reducing losses and increasing efficiency do not translate across sectors. Valuing water can also not be disaggregated from time, location, quality and precedent conditions; spatial and temporal distribution further complicates this task. For example, sustainability at the national level may not reflect sustainability at local or regional scales. Accounting problems are further complicated by the fact that many water uses are non-consumptive, linked to water quality, and difficult to quantify in natural landscapes due to a series of complex relationships, involving soil moisture, evaporation rates, microclimatic conditions, and so on. We rarely know the precise relations ship between:

- Aggregate water availability and physical output;
- Marginal relationship between water availability and economic output;
- Price to be assigned to non-financial impacts of water use and consumption.

The National Sustainability Indicator Program (NSIP), announced as a measure under the *Sustainable Australia - Sustainable Communities: a Population Strategy for Australia* in the 2011-12 Budget, aims to use the environmental accounting information to provide regular reporting on sustainability. It measures indicators related to human and social capital, natural capital and economic capital, as well as a number of contextual indicators. The indicators of the program relevant to aquatic ES are provided in Table 1 below:

Table 1: NSIP categories and indicators.

Category	*Indicator*
Land, ecosystems and biodiversity	• Extent of native vegetation • Ground cover • Ecosystem protection (protected areas)
Water Quality	• Water consumption • Water availability to meet demand
Natural Resources	• Fish stocks • Timber resources • Mineral and fossil fuel reserves
Contextual	
Land use	• Land use change

While environmental accounting has been identified as necessary to inform better management of our resources (including our natural resources), the current scope as exemplified in this table is by no means a guarantee that adequate information will be collected at the appropriate spatial and temporal scales. Large information gaps still exist, as will become evident from discussing the Murray-Darling Basin.

5c. South East Queensland ecosystem framework in local government.

Some important work, based on the Millennium Ecosystem Assessment (2005), was undertaken to develop the South East Queensland Ecosystem Services Framework (Maynard et al. 2010). The framework was developed with direct participation from experts, stakeholders and the community through a series of workshops, to make it relevant to regional stakeholders and maximise future adaptation in local planning and management processes. It consists of four main components:

1. Ecosystem Reporting Categories (ERC); 32 (ecosystem types)
2. Ecosystem functions (EF); 19
3. Ecosystem Services (ES); 28
4. Constituents of human well-being

Lack of sufficient quantitative data necessitated the use of a multi-criteria analysis where interactions between the four components were scored on a scale of 0-5 (none – strong). Three matrices were developed, to explore the relationships between the four components. Matrix 1 combines ERCs with EF (32x19); matrix 2 combines EF with ES (19x28), while matrix 3 multiplies the scores from matrix 1 and 2 to obtain constituents of human well-being. Work is proceeding on introducing different value weights. The ERCs and EF were also mapped in GIS (32 and 19 maps respectively), and overlays were also produced (one map combining all ERCs and one map combining EFs). Work is progressing on mapping ES. The conclusions the authors draw from the project are:

- Constructing a framework based on the Millennium Ecosystem Assessment is possible;
- Expert knowledge is necessary to fill data gaps, to provide local level knowledge and for maximal acceptance of the final product;
- Agreement on definitions, classification and general structure is essential;
- There are limits to the level of detail that can be achieved; this is a broad brush approach;
- Matrices can be used to comprehensively assess ecological attributes;
- Generating maps provides a first cut prioritisation of areas based on ecological significance;
- This work was supported by Agenda 21 for adoption at the national scale through the National Ecosystem Services Strategy (NESS).

The project, and the subsequent adoption for inclusion in the local government planning process (Dewar 2012) highlights that full integration into existing policy, planning and management strategies is complex and will require dedicated effort.

At the local government scale, the study found that relying on council natural resource management spatial data layers presents a 35% risk of inaccuracy in defining ecosystem functions. Dedicated resources are required to comprehensively integrate high ecosystem function value maps which are required to identify uses, set priorities and assess degree of fragmentation. Further, workshops are needed to link ES of local communities to political, scientific and community views for integration in council planning.

At the national scale, a mismatch exists between this work and the work of the National Water Commission (2012), which was developed at the same time. Whereas the SEQ ES approach is practical and applied, the NWC's benefits table remains a theoretical tool that provides a much less comprehensive framework. In addition, funding remains a big problem at national and state government level, and without it, many valuable frameworks such as the SEQ ES approach cannot be extended to other catchments and sub-catchments.

5d. The Murray Darling Basin

A study by CSIRO (2012) discusses ES and economic benefits in the context of restoring aquatic ecosystems in the Basin. It identifies freshwater (for drinking) and habitat for biodiversity as the most recognised ES by the community. Benefits from ES are direct and indirect; the former can be valued but considerable challenges remain. Comparing the cost of environmental flows (estimated at $542m annually) with the value of restoring floodplain vegetation, waterbird breeding, native fish and improving the Coorong, Lower Lakes and Murray Mouth (estimated to be worth between $3b and $8B), it is clear that there are considerable benefits in allocating more water for the environment. Other estimated values (that cannot be added due to the risk of double counting) are carbon sequestration ($120m - $1b), aesthetic appreciation ($330m), tourism ($160m) and avoiding damage and treatment of freshwater ($30m).

The concept of ES has been applied in the Murray-Daring Basin through determining *key environmental assets, key ecosystem functions, productive base and key* (estimated) *environmental outcomes* as part of environmentally sustainable level of take (Water Act 2007 s4[1]). Here, the debate has focused mainly on the over allocation of water for consumptive use and the need to restore the balance for environmental flow allocations through long term sustainable diversion limits (SDL).

The Basin Plan, which has recently been approved in Parliament, aims to return 2,750 GL of surface water by 2019 to the environment, a number that was derived from modelling environmental flow requirements necessary to achieve local environmental targets for 122 key environmental sites (hydrological indicator sites). These sites reflect the need for sustainable water use through key environmental flows at Basin and catchment scales, and key environmental assets (Murray-Darling Basin Authority 2012). *Key environmental assets* were defined using 5 criteria (Murray-Darling Basin Authority 2011):

- The water-dependent ecosystem is formally recognised in, and/or is capable of supporting species listed in, international agreements;
- The water-dependent ecosystem is natural or near-natural, rare or unique;
- The water-dependent ecosystem provides vital habitat;
- The water-dependent ecosystem supports Commonwealth, State or Territory listed threatened species and/or ecological communities;
- The water-dependent ecosystem supports or is capable of supporting significant biodiversity.

The number of wetlands occurring in the Basin is estimated to be around 30,000. There is no comprehensive list so further work will be required to comprehensively assess all of those. At present, the list of wetlands used is based on: Ramsar declared wetlands; wetlands listed on the Directory of Important Wetlands; High Conservation Value Aquatic Ecosystem sites; Icon sites established under The Living Murray Initiative; published and unpublished literature (e.g. jurisdiction management plans); and spatial data bases (e.g. the EPBC Act 1999 threatened species spatial layer).

The preliminary inventory identified over 2400 assets based on these criteria, for which insufficient information is currently available to determine environmental watering requirements. The indicator site approach was therefore developed, which assumes that a good spatial representation of indicator sites is the best interim solution to ensure environmental watering requirements for key environmental assets are met. A list of 14 *key ecosystem functions* was determined based on how they could be best influenced by environmental flows within the current knowledge and time constraints. These functions were also linked to the Functional Process Zones, established by Whittington et al. (2001), and allowed the hydrologic indicator site approach to be used to determine water requirements for key functions. The *productive base* is not defined by the Water Act (2007), but broadly it equates to ensuring that ES are supported (Reid-Piko et al. 2010); the terms are therefore treated synonymously.

The Murray-Darling Basin Authority (MDBA) considers that by providing environmental water requirements for ecosystem assets and functions, productive base requirements will also be supported, given that the ES concept is still in development and the relationship between biodiversity and ES is not sufficiently understood (Murray-Darling Basin Authority 2011). Environmental outcomes are defined under the Act to include ecosystem function, biodiversity, water quality and water resource health. Outcomes are defined at the Basin scale and for each of the 18 catchment regions of the Basin. They are related to beneficial outcomes for native fish, waterbirds and water dependent vegetation communities, and to the Ramsar listed wetlands such as the Murray mouth, Coorong and Lower Lakes and the icon sites along the Murray.

Most ecological targets are defined in terms of providing opportunities (e.g. recruitment for fish) based on flow regimes for which flow indicators have been developed; they are not (yet) quantified in terms of outcomes for those ecological groups. An ecological monitoring framework is also yet to be established. Environmental water has already been bought back on the water market through voluntary acquisitions by the Commonwealth Environmental Water Holder. Environmental watering plans are currently being developed, to optimise environmental flow releases to meet the targets as defined in the Basin Plan.

6. Conceptual framework (the way forward)

6a. A proposed ICEAIMM framework for integrating the ES concept with conservation, restoration and protection of freshwater ecosystems.

Implementing the ES framework requires a clear link to ecosystem restoration, conservation and protection of freshwater ecosystems for it to be meaningful as a tool for ecosystem protection. While ES enable us to quantify extent and nature of services, assess its level of sustainability, internalise the cost of degradation and define which trade-offs are desired, we need further measures to protect ecological capital to restore sustainable management of our freshwater resources. The Chinese SENCE model (Figure 1) recognises that human services are embedded in a larger bio-physical system (Wang & Paulussen 2007); this illustrates our ultimate dependency on vital services such as freshwater for our well-being, and hence the need to protect them for present and future generations.

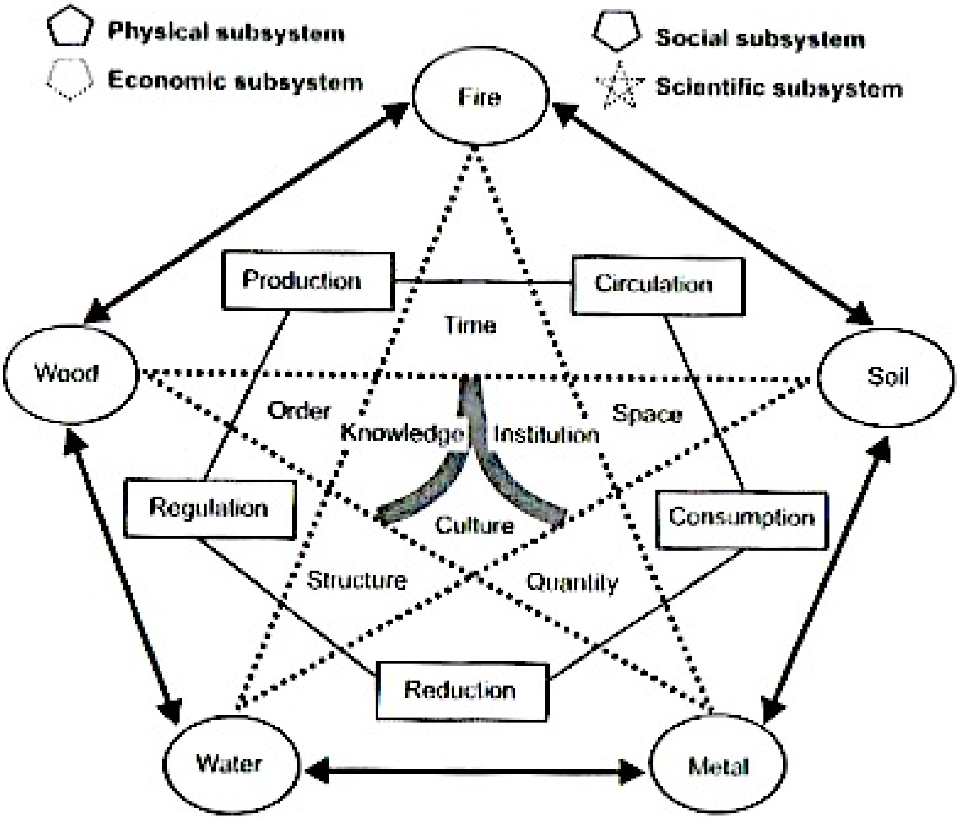

Figure 1. Social-Economic Natural Complex Ecosystem (SENCE) and its sustainability dimension (adapted from Wang & Paulussen 2007).

In this section, I propose a framework that is based on collective visioning of quantity and quality of services at the catchment scale by beneficiaries, experts and decision makers. The framework requires stakeholder participation and ownership, but also needs a solid foundation in science and be fully integrated in existing governance structures. The framework integrates a number of principles that were highlighted by various authors:

1. Trade-offs on ES to be defined in a participatory way, to ensure beneficiaries are involved in the decision making process (National Water Commission 2012a; Maynard et al. 2010; Dewar 2012)
2. Targets for restoration, conservation and protection of the freshwater ecosystem need to be defined in a participatory way and linked to adaptive management plans (Everard et al. 2009; Yung En Chee 2004)
3. A logical framework approach, where the relationships between each of the steps is made explicit (IWC 2010; National Water Commission 2012)
4. Ability to create incentive schemes (PES and EC) for conservation and restoration of ecosystem functions.
5. A decision support system for integrated solutions for river rehabilitation (Hermoso et al. 2012).
6. Capacity to integrate outcomes at different spatial scales, including building on national and international accounting methods, in order for

alignments between national and global initiatives to occur (UNU-IHDP & UNEP 2012; Millennium Ecosystem Assessment 2005; ABS 2012)

7. Provide opportunities to generate studies that link changes in biodiversity with ecosystem functions and changes in human well-being (Mace et al. 2012) as part of an adaptive management cycle.
8. Build upon existing knowledge systems and governance arrangements and aim at communities across ecosystems and sector boundaries (Primmer and Furman 2012).
9. Spatial heterogeneity (Zhang et al. 2010) and a more detailed local level analysis to complement measuring and mapping at larges scales (Primmer and Furman 2012).
10. Principle of subsidiarity (decisions are made at lowest possible levels) (White 2011).
11. Non-linear ecosystem responses require an adaptive management framework to periodically re-evaluate assumptions on pressures, restoration investments and so on (Holling & Gunderson 2002).

The proposed framework, presented in Table 2 below, consists of seven stages: 1.Informing, 2. Conceptualising, 3. Eco-linking, 4. Accounting/Trade-off, 5. Investing, 6. Managing and 7. Monitoring/Evaluation (ICEAIMM).

Table 2: The ICEAIMM framework, designed to integrate ES and ecological restoration projects in a management framework.

Stage	*Function*	*Stakeholders/providers*	*Description*	*Potential and available tools*
1. Informing	Data gathering and statistics, indicators, reporting, automated data production, similar to weather bureau data.	Government agencies and departments (e.g. Bureau of Meteorology in Australia, or Department of Environment in China)	Classifying the main ecosystems present in a country, and the quantity of this natural capital that is available. Indicator of quantity, to be standardised on a national/international scale. Environmental accounting tools are required.	National statistics, environmental accounting; international datasets, remote sensing; data from existing (and ongoing) monitoring programs.
2. Conceptualising	Identification of key ecosystem functions of each of the classified categories and their interactions (from step 1)	Interdisciplinary researchers, with a particular focus on ecological research; evidence based management experience and social learning.	Identifying the main ecosystem functions that we assume need to be maintained to keep the ecosystems in good condition, based on best available information/science. This needs to be done in an integrative way across all ecosystems (terrestrial, aquatic and marine), as the functions often link different (open ended) systems.	Social learning tools Bayesian Belief Networks Eco-hydrology modelling tools Existing ecological monitoring programs and studies
3. Eco-linking	Identifying socio-economic dependencies on ES at 'nested' spatial scales	Consultation between key stakeholders/beneficiaries, government (managers) and researchers.	Identifying the main ES that are derived from these ecosystems and their current demand. Includes potential future ES as well as those currently used. If potential ES cannot be identified, the precautionary principle should be applied. The less tangible ES here need to be identified as adequate conservation of representative	State indicators and performance indicators (Braat & De Groot 2012) Benefits table (NWC 2012) (=LFA) Social learning tools Bayesian Belief Networks Participatory Rural

			biodiversity and ecosystem conservation areas (public non-exclusive uses).	Appraisal tools (PRA)
4. Accounting/Trade-off	Environmental accounting (demand/supply balance).	Consultation between key stakeholders/beneficiaries, government (managers) and researchers.	Identifying areas with conflicting interests or which are out of balance (demand>capacity, or supply>demand). At this stage the requirements to maintain or restore an ecosystem need to be quantified, and areas for trade-offs need to be earmarked.	Accounting and modelling tools, GIS, remote sensing imagery, information from previous stages.
5. Investing	Restoration and conservation management programs.	Top down planning approach, requiring government commitment but also buy in on the ground	Identifying & prioritising restoration programs, trade-offs. Providing security to all stakeholders, including the environment, using decision support systems.	Existing ecological restoration programs (ERP) can be evaluated in the light of identified requirements.
6. Managing	Management plans are worked out in detail and implemented by a range of participants	All relevant participants (includes engineers, hydrologists, managers, CMAs, volunteers,…)	Engineering and operations programs, integration with restorations and conservation programs.	Adoption of and rationalising existing ERPs; existing tools and information as for previous stages
7. Monitoring and evaluating	Monitoring, evaluation and adaptive management	Managers, data analysts, key program leaders	Monitoring program should collect regular information on ecological condition and trend, and on the use of ES. At a predetermined frequency, this information should be analysed and evaluated; steps are described in more detail below.	Existing monitoring programs may be looked at to see if they can be used or provide baseline data.

1. The informative stage corresponds to the various attempts of governments to 'take stock' of the current environmental assets. It is important that this stage develops adequate measurable indicators that allow standardised comparisons across the range of ecosystems that have been classified and quantified in terms of extent. The environmental accounts should also be compatible with the international ES framework, and with the 'State of the World' annual reports issued by the Worldwatch Institute in the last couple of decades. There will be a balance between developing an accounting framework that is adequate for the purposes of national, regional, catchments and local assessments, and compatibility with international classification systems.
2. In the conceptualising stage, identifying the main ecosystem functions that need to be maintained to keep the ecosystems in good condition, based on best available information/science will require bringing together information and studies from various sources on different ecosystems. Ideally, this task should start from the information provided in step 1, and zoom in on more detail as required. Dewar (2011) provides a good example of how this is done at the regional and local level in South East Queensland. For example, if the national environmental accounting system classifies a number of terrestrial, aquatic and marine ecosystems, their key functions can be examined initially. The interlinking between those will then require researchers to focus on ecotones (for example floodplains and riparian vegetation between terrestrial and aquatic ecosystems, mangroves between terrestrial/aquatic and marine ecosystems). Even if there is insufficient information available on the interlinking functions, it will be important to identify these in the most comprehensive manner. They will become crucial in terms of management and trade-offs, as human consumption may leave the remaining capacity unable to maintain functionality for one or more ecosystems.
3. Eco-linking here stands for linking ecology with economy. The task requires information on a wide array of economic information collected in Step 1, including current landuse, permits, size of cities and average quantities of water supply, hydroelectricity, agriculture, mining, aquaculture and silviculture. The analysis requires an examination of all ES people depend on that originate in their country and their current demand. This requires engagement from government with the key actors (stakeholders) to understand current dynamics and links and includes potential future ES that are known to come online in the near future. If potential ES cannot be identified, the precautionary principle should be applied (e.g. high biodiversity is valuable, even if potential value is currently unknown). Key actors can be defined as those who rely on ES for their livelihood (subsistence or income generating reliance) and their professional representation organisations or those for whom a particular service is essential (e.g. drink water supply). Identifying key stakeholders is as challenging as identifying key ecosystem functions and services; some simplifications are unavoidable if we want to commit to some management implementation. The less tangible ES here need to be identified as adequate conservation of representative biodiversity and ecosystem conservation areas (public non-exclusive uses).

4. At the accounting/trade-off stage, the demand and the supply capacity of the systems are quantified and balance sheets are created. An important part of this step is to quantify also what becomes a net export of economic products, for which reliance on an ecosystem service was necessary in the production process. This needs to be accounted for by comparing it to imports, and will require an economic evaluation; most of this information is available from national economic statistics. Trade-offs of ES will to varying degrees impact on both ecosystems and on beneficiaries. The type of dependency on ES may be considered in order to weight its importance, whereby a subsistence farmer or fisherman would receive a higher weighting than a transnational corporation, because even though in terms of net contribution to GDP the farmer contributes less, his greater vulnerability would invoke an equity principle. A problem that may arise at this step is in how one quantifies carrying capacity of ecosystems, i.e. its ability to supply a certain amount of a particular service in a sustainable way. Ecosystem degradation is usually a sign that demand outstrips supply capacity; which results in natural capital being eroded. Extrapolation into the future will be useful to project scenarios in perhaps 10 years time, and start planning to address future demand pressures.
5. The investment stage is the point where commitment from the government will be demonstrated. If demand outstrips supply, and degradation has been verified, restoration and conservation programs will be required. Demand pressures may need to be reduced (through efficiencies, changing policies, deterrence and incentives) unless a clear trade-off decision has been reached. Alternatively, ecological capital may need to be increased, by setting clear management targets to restore condition or improve/increase quantity of the ecosystem (for example by building corridors). Governance will play a strong role here. Market valuation is required, but only to compare value of benefits received versus investment required to maintain or restore sustainable ES. The value of the ecological capital is not required (often estimated to be much higher than market value and thus unsuitable in comparisons); only the marginal value to maintain or restore the capital. Equally, we do not put a market value on human capital, but we use market valuation to invest in it (e.g. education).
6. In the management stage, detailed management plans are worked out and implemented. Many ecosystems are already modified due to human interactions, and can be maintained in a modified state. But this requires active management and there are optimal ways of prioritising and doing this, including restoration activities where required. Decision support tools, which include quality trade-off options, should be used in consultation with stakeholders to develop an optimal outcome (Hermoso et al. 2012). As a principle, if ES can be used by multiple users, this should get preference over exclusive use by a few. The environment may have multiple users as well, such as fish, vegetation, birds and many others. Examples are dams in rivers, which require sedimentation management, fish passage, flow release management for consumptive use and environmental flow release, among others. Governance here will include interactions between key stakeholders

and government, to provide two perspectives: on-the-ground management perspectives as well as catchment management and wider regional and national policy perspectives, including restoration programs, funding and institutional support mechanisms. The Australian CMA model may best fit this requirement, but needs to be critically revised, as current CMAs are not necessarily representative of all key stakeholders.

7. Monitoring and evaluating (adaptive management stage). This step involves:
    a. Periodic checking of environmental accounting statistics. Where are the trends heading? Do they reflect intended targets, or are restoration/conservation attempts failing? Are our measurements at the right spatial and temporal scales, or is there vital information lacking? This step should be undertaken by government, and reported to the stakeholders/beneficiaries, and to the general public.
    b. Evaluating the main ecosystem functions that were identified. This is an area of uncertainty where hypotheses need to be formulated based on available information and expert advice. Periodically, new evidence will turn up that may point in a different direction. Sources of evidence are obtained from social learning, research and evidence based management experience. If quantitative hypotheses are formulated, the evaluation can be used to try to reduce the level of uncertainty attached to it.
    c. Evaluating the main ES that are derived from the range of ecosystems present. Are they still the same? Has demand for the services increased, decreased, changed in spatial/temporal distribution? What is the economic value that it represents, and how does this compare against investments to maintain and/or restore natural capital? What would be the alternatives, if the investment cannot be maintained? What are emerging demands? How do they project into the future? The assumptions about key beneficiaries (stakeholders) may need to be revisited or validated. Are they still the same or have some new ES demands emerged? From the same or different groups?
    d. Is demand more balanced against supply capacity of the ecosystems? Here, the assumptions about ecosystem capacity to supply ES, and resilience against demand pressures need to be evaluated. Has new knowledge contributed to a greater insight into these ecosystems? Do we know more about potential tipping points, or ways we can measure resilience? Have some ecosystems tipped to another equilibrium, which has influenced the capacity for ES delivery? Has this reduced its capacity, or changed the type and quantity of ES available (the integrative assessment across ES is an important part of this evaluation).
    e. Evaluating restoration programs. Have the targets that were set at the start of the program being met, or are we on track of meeting them by the projected timeline? If not, what are the reasons why? Investment failure? Are there other drivers that were not accounted for? Do we need to adjust the targets and are they still relevant (see d)? Do we need to increase our restoration efforts, or revisit the

overall strategy (adaptive management)? How well does the restoration programs work at the local, catchment, Basin and national scale?

f. How effective is our monitoring and evaluation program? Does it inform success or failure, according to the original objectives? Given that we manage complex human-ecosystem interactions, what are the uncertainty factors that we were unable to capture? Do we monitor at the appropriate spatial and temporal scales, and do we include the relevant stakeholders? Have we captured relevant cross-sectoral information, and are our governance models adequate to deal with the challenges? Ideally, the monitoring should reveal that revisiting the cycle 1-6 should make the program more effective, and this can be measured also by the amount of effort (time?) spent on effectiveness monitoring. If this decreases or stays the same at each cycle, then the indicator suggests that the monitoring and evaluation program is effective. If it increases, this may be an indication that our assumptions were wrong, and that the management program needs significant adjustment and redesign. Given that this is an adaptive management framework, decreasing effectiveness should not be viewed as failure, but as an opportunity to learn and improve our management practices.

Planning, monitoring and evaluation should constitute only 10% of total ERP investment, but it is important that a long term commitment is made to invest this money, as it will be crucial to evaluate performance, efficiency and effectiveness of public moneys spent. Monitoring and evaluation should involve key partners and information from all previous stages.

6b. Example of how ICEAIMM can be used in the Yellow River, China.

The Yellow River is trying to manage delivery of four ES: water provision for drinking water and irrigation, environmental flows to restore ecosystem health including water for the wetlands in the lower delta, assimilation of wastewater, and sediment management to reduce the risk of flooding.

1. In the *information* stage, all information which is currently vested in different government departments and research institutions will need to be brought together, standardised, and developed into indicators that are informative about the extent of the ecosystems existing along the Yellow River. This stage needs to tie in with the development of national classification systems that can accommodate an inventory of all existing ecosystems, and with the international work on ES, and should focus on information relevant to water allocation and modelling of capacity of flows to meet the ES demand.
2. In the *conceptualisation* stage, ecosystem functions need to be identified and integrated. One example here is to develop in-stream indicators that complement those of the SLCP program, which was intended to reduce sedimentation from the Loess plateau, as discussed earlier. Another example is the environmental flow allocation as discussed by Shen and Speed (2009)

to ensure appropriate flushing of sediments in the lower reach, and combining this with ecological watering regimes for the wetlands in the delta.

3. At the *eco-linking* stage, the river demands are linked to the ecosystem functions. At this stage it is possible to establish a Bayesian Belief Network (BBN) model, where the nodes are linked together and the model is set up for quantitative inputs at the next stage. Here, the environmental flow demands can be modelled against water demands for irrigation and drinking water use, and a number of optimisation scenarios being created, to be further populated with quantitative data at the next stage.
4. At the *accounting* stage, the BBN model can be populated with data obtained from stage 1, according to scenarios derived from stages 2 and 3. The model can deal with uncertainty and lack of data for some components, so that approximations can still be derived. In the absence of data, quantitative estimates can be used that may be obtained from interpolation (e.g. using comparable data from similar catchments elsewhere). The aim here is not to get very precise information, but to discover where the areas of discrepancy are between demand and ecosystem capacity, under current and perhaps a number of hypothetical scenarios.
5. *Investment* decisions can be made based on the findings from stage 4. In our example of the SLCP program, once all critical factors have been modelled (drivers, priority areas for revegetation, high erosion areas and others) it becomes possible to define rehabilitation actions and estimate the associated costs, based on priority areas. The water allocation regime can also determine the level of investment in complementary measures for restoration of the delta wetlands, or the investments required for infrastructure works to complement flushing regimes.
6. Once investment decisions are made, *management* plans are required for the areas which will be rehabilitated, conserved or used in different ways. Each plan will need to take into account existing management initiatives, to assess how they can be improved to meet determined management objectives.
7. *Monitoring and evaluation* will need to set out a comprehensive plan with integrated and nested spatial objectives, to which suitable indicators are linked. Environmental flow indicators will need to be coupled to sedimentation and water quality measures, which in turn need to complement erosion indicators that may already exist for a program such as the SLCP program.

6c. Example of how ICEAIMM can be used in the Murray-Darling Basin, Australia.

In the Murray-Darling Basin, the Basin Plan has defined sustainable diversion limits, as a first step to restore river health. However, there is still an enormous challenge in designing environmental watering plans to optimise watering regimes for vegetation, fish and birds, at three nested spatial scales: the Basin scale, the catchment scale and the ecological asset scale. This will then also need to be coupled to monitoring plans that can measure the effects of environmental flows at short, intermediate and longer term timescales. In addition, the ecological effects of water trade, which will increase water use efficiency (including buy-backs for environmental watering), are yet to be assessed and incorporated into restoration efforts.

1. The MDBA is currently informing itself about environmental accounting initiatives undertaken by other government departments in order to fill substantial gaps in *information* on ecological assets. It is unlikely that these gaps will be filled soon, nor will there be sufficient detail to undertake an ecosystem analysis similar to the SEQ ES framework. More likely, MDBA will need to refine its current method of inferring adequate ecological health management initiatives such as delivery of environmental flows by linking outcome monitoring to increased information on ecological types, functions and assets. However, the Basin Plan will not come into effect fully until 2019, so this lead up period could be used to undertake data gathering through partnerships with other Australian government departments. All indications are that this is currently being planned for.
2. *Conceptualising* will need to link and integrate the current ecological flow concepts with other important drivers of ecological health and with the effects of water trade as they become apparent. For example, Pittock et al. (2012) pointed out the new externalities created between water and forest mechanisms; this is the result of considering water ES in isolation from terrestrial ecosystems, where they should clearly be assessed in an integrated way based on a national classification scheme for all ecosystems in the Murray-Darling Basin. MDBA may not be the best institution to drive this integration, but could well advocate such an integrated approach to be pursued by the Department for Sustainability, Environment, Water, Population and Communities and the National Water Commission.
3. Similar to the Yellow river, *eco-linking* needs to tie water demands for irrigation of crops, grazing areas, forests, and drinking water purposes to environmental flow demands. This work has been done under the Basin Plan at the Basin scale, but will require further resolution at smaller spatial scales, with regard to environmental flow allocations. It could be tied in with the MDBA's commitment to localism, i.e. consulting with local communities to ensure local knowledge will be included in the implementation of the Basin Plan. Further work will be required in modelling the effects of environmental flows on ecological outcomes.
4. *Accounting* will require a more comprehensive database derived from Stage 1. The information required for this stage will need to be planned ahead and may require specific investments by MDBA to 'fill the gaps' with regard to what can be harvested from other government departments. It includes baseline information on current condition and demand for ES (in particular consumptive water demand). However, other demand factors may put pressure on the water resources (land use and infrastructure changes, for example), so the need for integrating the Murray-Darling Basin ES framework within a national (and international) framework is emphasised once again.
5. *Investment and restoration planning.* Some of this is already done through water buy-back schemes and implementation of the Basin Plan. However, as the Basin Plan is envisaged to be implemented as an adaptive management

cycle, the need will increase to integrate the plan with other natural resource management initiatives (state and federal) at the catchment management authorities' level. The investment budget to implement the Basin Plan and complementary natural resource management initiatives will still need to be secured through intergovernmental agreements.

6. *Management plans* will need to capitalize and build on existing management frameworks. However, these are in the process of being revised now that Basin Plan implementation is imminent. Existing arrangements reflect consultation with state working groups around natural resource management programs and there is a need to build more integrative consultation processes, which will reflect complementary functions rather than standalone initiatives. These will include water allocation, environmental flow regimes, works and measures to remove flow delivery constraints, water efficiency infrastructure as well as existing initiatives such as salt interception schemes, fish ladders, and the like. A stronger emphasis on capacity building with local on-ground NRM organisations such as CMAs will be crucial for achieving on-ground results.
7. *Monitoring programs* are currently in a conceptual stage, and will require greater clarity in governance arrangements. At present, the Water Act (2007) would suggest that MDBA is responsible for monitoring at the Basin scale, and perhaps at the catchment scale (as some catchments cross state jurisdictional boundaries). The assets would be the responsibility of state governments. But some assets also share state jurisdiction boundaries, and there are many issues yet to be clarified, not least funding agreements between MDBA and the state partners for monitoring programs.

6d. SWOT analysis of the proposed ICEAIMM framework

The proposed framework will certainly not be a panacea for integrating ES with river restoration programs. We already concluded that in order for the ES framework to work in an effective way, it needs to be complemented and integrated with ecosystem conservation and restoration frameworks, at multiple spatial and temporal scales. ICEAIMM is an attempt to do this, and it is therefore valuable to analyse the potential strengths, weaknesses, opportunities and threats of this framework.

Table 3: SWOT analysis of the ICEAIMM framework

Strengths	***Weaknesses***
1. Integrates many of the ideas proposed by prominent authors as discussed in this essay. 2. The seven stages allow for flexibility in application to specific river basin situations, as applied here for the Yellow River and the Murray Darling Basin. 3. Integrates ES quantification with	1. No clear link is provided to the international framework and classification work that exists at that scale. 2. The framework is case specific (conceptualising, eco-linking, and so on) and has the potential to misalign if integration at larger spatial and temporal scales is attempted as a

restoration and conservation of the same ecosystems that are threatened. 4. Integrates across sector and ecosystem divides. 5. Can be applied at a number of different spatial and temporal scales; hence has the potential to integrate across those scales. 6. Identifies ES in a participatory process with stakeholders and beneficiaries. 7. Focuses on the process, including the adaptive management cycle.	retrofit. 3. The process does not explain how integration across spatial and temporal scales should be done (top down, bottom up), nor what the appropriate delineations of the ecosystem units are to be considered for assessment. 4. The framework does not address issues of eco-compensation or payment for ES in an explicit way; these mechanisms are to be linked to stage 5 (investment). 5. The process of how to identify beneficiaries to be involved is not described.
Opportunities	***Threats***
1. In Australia, the ES approach can be applied at a national scale, as part of the National Water Initiative, and linked to the Millennium Ecosystem Assessment. An implementation framework could use the NWC's benefits table, and build on the work from Maynard et al. (2010). This could put Australia at the forefront of the ES approach in demonstrating how a global framework can be implemented at local scale in an integrative way. 2. In China, the advance on eco-compensation and payment for ES provides a starting point for applying the framework in the area of strategic restoration and conservations investments. Chinese expertise in this area is valuable to apply elsewhere, and will contribute to the global framework.	1. The proposed framework is unknown, and may not get sufficient exposure to be accepted by intended audiences for uptake and implementation. 2. The framework tries to span too many efforts across water resource management, and unless the socio-political governance constellation is favourable, it may not be adopted without clear champions. 3. Implementing the framework across all geographic areas may be seen as too costly, so further work on financial priorities and commitments may be required.

As with all frameworks, the benefit lies in bringing a number of key elements together to assist with a more successful implementation of the ES approach, which is a relatively new concept in both Australia and China (and in fact around the world).

7. Conclusion and recommendations

While Primmer and Furman (2012) recognize that measuring all ES is unrealistic, it is still possible and desirable to quantify ecological capital on a global scale. This will help us establish international treaties and protection mechanisms to ensure we arrest the global decline of some of the key regulating ecosystems. In their paper on planetary boundaries, Rockström et al. (2009) point out that we cannot continue to loose biodiversity at the current rate; this means drastic action at a global scale will be required. The Millennium Ecosystem Assessment (2005) projects four future scenarios, where human well-being is quantified for five aspects: material well-being, social relations, health, security and freedom of choice. For each of those, the loss of biodiversity is estimated. Two of the scenarios with a proactive approach to environmental management (techno-garden and adapting mosaic) result in least biodiversity loss, and would form a good starting point for developing a global strategy.

At different spatial scales, different priorities for quantifying ES prevail: at the global scale, the task is to identify those regulating ES critical to our survival as a species, as well as the critical amount of biodiversity necessary to preserve and maintain. At the national scale mainstreaming the ES concept into economic modelling is required to improve our ability to measure human well-being (Pennock and Ura, 2011). At a basin scale, the concept can be used to maintain the integrity of the aquatic ecosystem on a sustainable and equitable use basis. At the valley scale human-ecological coupled systems need to integrate current and future demand projections to achieve longer term sustainable use of our natural resources. At the reach scale, local knowledge from on-ground use and management of natural resources will inform the larger scale planning processes, and will become the 'hands, eyes and ears' of achieving restoration targets defined in a broader adaptive management cycle.

PES and EC are concepts that have the potential to be used in an international context, as the need to act on a global scale is becoming more urgent. As we are moving towards a more sustainable economic framework, these concepts should be integrated in the SEEAW frame work (UNU-IHDP & UNEP 2012) and could be linked to debt for nature swaps which have been around for the last three decades (Sheikh 2010). China is uniquely placed to contribute its experience in this area at a global scale. Australia has already adopted the SEEAW framework (ABS 2012), and MDBA, by being positioned at the Basin-scale level, has a unique governance opportunity to further link national policies and frameworks to the local scale with assistance of the states and CMA partners. Therefore, Australia has the potential for becoming the first country to demonstrate how the international ES framework, as defined in SEEAW, can be applied at national and smaller spatial scales. This is important, because it provides an opportunity for integration of environmental accounting from local to global scales.

For Australia and China, decisions about knowledge exchange should be made as part of national priority settings. The key policy documents that provide opportunities for this are the No.1 Policy document, released in China in 2011, and the National Water Initiative (2004), Australia's long-term policy for water reform, which aims to 'optimise social, economic and environmental outcomes' (National Water Commission 2012). For the Murray-Darling Basin, the Water Act (2007) and the Basin Plan (2012) provide further opportunities.

This study has explored the effectiveness of ES as a framework for protecting ecosystems, in the context of China and Australia. Knowledge exchange opportunities have presented themselves through the five year ACEDP project for China and Australia to learn from each other, and to identify where future opportunities lie in continuing this dialogue. Three key recommendations have emerged:

Recommendation 1:
ES should be linked explicitly to restoration and conservation frameworks, in order to integrate the strategy of 'stock taking' and valuation of ecological capital with strategies to preserve and maintain this capital.

Recommendation 2:
The proposed ICEAIMM framework provides a mechanism to integrate ES with ecological restoration and conservation. It is recommended to explore the use of this framework to implement Recommendation 1.

Recommendation 3:
Dialogue should be maintained between Australia and China about water resource management, to foster opportunities for knowledge exchange, and to work jointly on integrating national ES projects under the international framework as proposed by the Millennium Ecosystem Assessment.

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Appendix 1: example of a beneficiaries table (after National Water Commission 2012a).

Beneficiaries	Benefits	Change in benefits	Service	Change in service	Current water regime	Water regime under allocation	Other processes	Condition of other processes	Risk to service
A	A	B	A	B	B	B	A	B	A
A. Together with stakeholders, identify beneficiaries, benefits, services and processes to fill in cells (A) from left to right									
B. Together with scientists and experts, show how changes in water regimes affect benefits, fill in cells (B) right to left									
Beneficiary 1	Benefit 1	Change 1	Service 1	Change in service 1	Water regime responsible for service 1	Water regime responsible for change in service 1	Other processes 1	Condition of other processes 1	Risk to service 1
Beneficiary 2	Benefit 2	Change 2	Service 2	Change in service 2	Water regime responsible for service 2	Water regime responsible for change in service 2	Other processes 2	Condition of other processes 2	Risk to service 2
Beneficiary 3	Etc…								

Note: there are not always one-to-one relationships between beneficiaries, benefits, services and functions (water regime).

Logical steps in filling out the beneficiaries table:

1. List all beneficiaries and describe the resource (A)
2. Determine how each benefit is dependent on services provided by the aquatic system (A)
3. Determine how each service is dependent on the water regime and other processes (B)
4. Determine how changes in water regimes might affect beneficiaries (B)
5. Setting high level objectives (A+B)
6. Informing trade-offs (A+B)