Report No. **187**

epa Research
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

ESManage Literature Review Ecosystem Services in Freshwaters

Authors: Hugh B. Feeley, Michael Bruen, Craig Bullock, Mike Christie, Fiona Kelly, Kyriaki Remoundou, Ewa Siwicka & Mary Kelly-Quinn

Comhshaol, Pobal agus Rialtas Áitiúil
Environment, Community and Local Government

www.epa.ie

ENVIRONMENTAL PROTECTION AGENCY

The Environmental Protection Agency (EPA) is responsible for protecting and improving the environment as a valuable asset for the people of Ireland. We are committed to protecting people and the environment from the harmful effects of radiation and pollution.

The work of the EPA can be divided into three main areas:

Regulation: *We implement effective regulation and environmental compliance systems to deliver good environmental outcomes and target those who don't comply.*

Knowledge: *We provide high quality, targeted and timely environmental data, information and assessment to inform decision making at all levels.*

Advocacy: *We work with others to advocate for a clean, productive and well protected environment and for sustainable environmental behaviour.*

Our Responsibilities

Licensing

We regulate the following activities so that they do not endanger human health or harm the environment:

- waste facilities *(e.g. landfills, incinerators, waste transfer stations)*;
- large scale industrial activities *(e.g. pharmaceutical, cement manufacturing, power plants)*;
- intensive agriculture *(e.g. pigs, poultry)*;
- the contained use and controlled release of Genetically Modified Organisms *(GMOs)*;
- sources of ionising radiation *(e.g. x-ray and radiotherapy equipment, industrial sources);*
- large petrol storage facilities;
- waste water discharges;
- dumping at sea activities.

National Environmental Enforcement

- Conducting an annual programme of audits and inspections of EPA licensed facilities.
- Overseeing local authorities' environmental protection responsibilities.
- Supervising the supply of drinking water by public water suppliers.
- Working with local authorities and other agencies to tackle environmental crime by co-ordinating a national enforcement network, targeting offenders and overseeing remediation.
- Enforcing Regulations such as Waste Electrical and Electronic Equipment (WEEE), Restriction of Hazardous Substances (RoHS) and substances that deplete the ozone layer.
- Prosecuting those who flout environmental law and damage the environment.

Water Management

- Monitoring and reporting on the quality of rivers, lakes, transitional and coastal waters of Ireland and groundwaters; measuring water levels and river flows.
- National coordination and oversight of the Water Framework Directive.
- Monitoring and reporting on Bathing Water Quality.

Monitoring, Analysing and Reporting on the Environment

- Monitoring air quality and implementing the EU Clean Air for Europe (CAFÉ) Directive.
- Independent reporting to inform decision making by national and local government *(e.g. periodic reporting on the State of Ireland's Environment and Indicator Reports).*

Regulating Ireland's Greenhouse Gas Emissions

- Preparing Ireland's greenhouse gas inventories and projections.
- Implementing the Emissions Trading Directive, for over 100 of the largest producers of carbon dioxide in Ireland.

Environmental Research and Development

- Funding environmental research to identify pressures, inform policy and provide solutions in the areas of climate, water and sustainability.

Strategic Environmental Assessment

- Assessing the impact of proposed plans and programmes on the Irish environment *(e.g. major development plans)*.

Radiological Protection

- Monitoring radiation levels, assessing exposure of people in Ireland to ionising radiation.
- Assisting in developing national plans for emergencies arising from nuclear accidents.
- Monitoring developments abroad relating to nuclear installations and radiological safety.
- Providing, or overseeing the provision of, specialist radiation protection services.

Guidance, Accessible Information and Education

- Providing advice and guidance to industry and the public on environmental and radiological protection topics.
- Providing timely and easily accessible environmental information to encourage public participation in environmental decision-making *(e.g. My Local Environment, Radon Maps).*
- Advising Government on matters relating to radiological safety and emergency response.
- Developing a National Hazardous Waste Management Plan to prevent and manage hazardous waste.

Awareness Raising and Behavioural Change

- Generating greater environmental awareness and influencing positive behavioural change by supporting businesses, communities and householders to become more resource efficient.
- Promoting radon testing in homes and workplaces and encouraging remediation where necessary.

Management and structure of the EPA

The EPA is managed by a full time Board, consisting of a Director General and five Directors. The work is carried out across five Offices:

- Office of Environmental Sustainability
- Office of Environmental Enforcement
- Office of Evidence and Assessment
- Office of Radiological Protection
- Office of Communications and Corporate Services

The EPA is assisted by an Advisory Committee of twelve members who meet regularly to discuss issues of concern and provide advice to the Board.

EPA Research Programme 2014–2020

Ecosystem Services in Freshwaters

(2014-W-LS-5)

ESManage Literature Review

Prepared for the Environmental Protection Agency

by

University College Dublin

Authors:

Hugh B. Feeley, Michael Bruen, Craig Bullock, Mike Christie, Fiona Kelly, Kyriaki Remoundou, Ewa Siwicka and Mary Kelly-Quinn

ENVIRONMENTAL PROTECTION AGENCY
An Ghníomhaireacht um Chaomhnú Comhshaoil
PO Box 3000, Johnstown Castle, Co. Wexford, Ireland

Telephone: +353 53 916 0600 Fax: +353 53 916 0699
Email: info@epa.ie Website: www.epa.ie

ACKNOWLEDGEMENTS

This report is published as part of the EPA Research Programme 2014–2020. The programme is financed by the Irish Government and administered by the Environmental Protection Agency, which has the statutory function of co-ordinating and promoting environmental research.

The authors wish to thank the following for their comments, steerage and advice: Dr Catherine Bradley, Mr Alexandre Bredimas, Dr Colin Byrne, Dr Donal Daly, Dr J. Iwan Jones, Mr Martin McGarrigle, Dr Marc Metzger, Ms Petrina Rowcroft, Dr Tom Stafford and Dr Alice Wemaere. The photograph on the front cover is courtesy of Dr Martin O'Grady, Inland Fisheries Ireland.

DISCLAIMER

Although every effort has been made to ensure the accuracy of the material contained in this publication, complete accuracy cannot be guaranteed. Neither the Environmental Protection Agency nor the authors accept any responsibility whatsoever for loss or damage occasioned, or claimed to have been occasioned, in part or in full, as a consequence of any person acting, or refraining from acting, as a result of a matter contained in this publication. All or part of this publication may be reproduced without further permission, provided the source is acknowledged.

The EPA Research Programme addresses the need for research in Ireland to inform policymakers and other stakeholders on a range of questions in relation to environmental protection. These reports are intended as contributions to the necessary debate on the protection of the environment.

EPA RESEARCH PROGRAMME 2014–2020
Published by the Environmental Protection Agency, Ireland

ISBN: 978-1-84095-665-8 September 2016

Price: Free Online version

Project Partners

Dr Hugh B. Feeley
School of Biology and Environmental Science
University College Dublin
Dublin
Ireland
Tel.: +353 1 716 2339
Email: hugh.feeley@ucd.ie

Professor Michael Bruen
School of Civil Engineering
University College Dublin
Dublin
Ireland
Tel.: +353 1 716 3212
Email: michael.bruen@ucd.ie

Dr Craig Bullock
School of Geography
University College Dublin
Dublin
Ireland
Tel.: +353 1 716 2784
Email: craig.bullock@ucd.ie

Professor Mike Christie
Blue Island Consulting Ltd
Ynyslas, Borth
UK
Tel.: +44 1970 622217
Email: blueisland_consult@btinternet.com

Dr Fiona Kelly
Inland Fisheries Ireland
Citywest Business Campus
Dublin
Ireland
Tel.: +353 1 884 2600
Email: fiona.kelly@fisheriesireland.ie

Dr Kyriaki Remoundou
Blue Island Consulting Ltd
Ynyslas, Borth
UK
Tel.: +44 1970 622522
Email: kyr2@aber.ac.uk

Ms Ewa Siwicka
Blue Island Consulting Ltd
Ynyslas, Borth
UK
Tel.: +44 74 8191 1997
Email: siwicka92@gmail.com

Associate Professor Mary Kelly-Quinn
School of Biology and Environmental Science
University College Dublin
Dublin
Ireland
Tel.: +353 1 716 2337
Email: mary.kelly-quinn@ucd.ie

Contents

List of Figures

List of Tables

Executive Summary

The benefits derived from the environment and nature have long been recognised by human societies. Since the 1970s, the concept of ecosystem services has evolved and it is now considered a means of embedding biological and ecological thinking into policy and practice. It is also seen as an effective means of communicating to all stakeholders, including the general public, the value of protecting our environment and its ecosystems, in order to maintain the flow of benefits (i.e. goods and services) that they provide to human societies. In the simplest of terms, ecosystem services are defined as "the contributions that ecosystems make to human well-being". The adoption of an ecosystem services framework in the management and protection of our freshwater ecosystems allows for the quantitative valuation (both monetary and non-monetary) of changes in specific ecosystem services and their flows, the protection of their biodiversity and the integration of the formerly separate disciplines of economics and ecology, in order to better represent and investigate the relationships between human economies and natural environments.

A plethora of terminologies relating to ecosystem services has, to some extent, complicated efforts to communicate the concept and its benefits to a wide range of stakeholders. Several classification frameworks [e.g. the Millennium Ecosystem Assessment (MEA), The Economics of Ecosystems and Biodiversity (TEEB) and the Common International Classification of Ecosystem Services (CICES)] have been developed in an attempt to organise and understand the stocks and flows of ecosystem services, and the goods and benefits that society can utilise. Four main categories of ecosystem services have arisen from these classification frameworks: (1) provisioning services; (2) regulating and maintenance services; (3) cultural services; and (4) a range of supporting processes (also referred to as supporting or intermediate services), which underpin the services of the first three categories. Provisioning services include water for drinking and non-drinking purposes (e.g. irrigation, cleaning, agricultural and industrial use), and the provision of food (e.g. fish). Regulating and maintenance services include those that directly (e.g. waste assimilation and pathogen control) and indirectly (e.g. regulation of decomposition, climate and flows) sustain environmental quality. Finally, cultural services include tangible recreational uses (e.g. kayaking, angling and walking along rivers) and also contribute to less tangible benefits, such as bequest, aesthetic or spiritual benefits, as well as having educational value. The sustainable delivery of these so-called "final ecosystem services" (i.e. provisioning, regulating and maintenance, and cultural services) requires a minimum level of ecosystem "infrastructure". This is why supporting processes (also referred to as "supporting services") and their connection to, and within, the physical environment are required to underpin the provision of other freshwater goods and services. In fact, by definition, ecosystem services must be underpinned by ecological processes and biology, and can be distinguished from abiotic (geosystem) services, such as mineral extraction, power generation and navigation. Similarly, the ecosystem services framework should not be confused with the "ecosystem approach" (as endorsed by the Convention on Biological Diversity), integrated water resource management or the water–energy–food nexus. While all four of these "approaches" have many common features in terms of their genesis, underpinning assumptions, and objectives and methods, especially in terms of the sustainable use of natural capital and ecosystems, they differ in their emphasis.

The specific details of the links between biodiversity, ecological processes and ecosystem services are largely unknown. Much of the evidence has come from studies of the relationship between biodiversity and ecosystem functioning, especially with regard to regulating and maintenance services. Nevertheless, the consensus in the literature is that biodiversity loss compromises ecosystem resilience and, therefore, the capacity to sustainably deliver ecosystem services. Furthermore, anthropogenic influences, such as changes to social behaviour, which in turn lead to shifts in social structures (e.g. population numbers), can change the demand on ecosystem services (e.g. clean water for abstraction), that is, changes in one system (e.g. the social system) will cause feedback effects on biodiversity and the functioning of the ecological system, and vice versa.

Importantly, the functions or processes of ecosystems become *ecosystem services* only if there are beneficiaries (i.e. humans that benefit from them). Therefore, beneficiaries drive active or passive consumption and/or appreciation of ecosystem services, which results in an impact (positive or negative) on their welfare. A knowledge of the location of the beneficiaries and the area in which there is a demand for ecosystem services is essential for the design of environmental management policies and the targeting of management interventions. Similarly, the values assigned to ecosystem services will vary depending on the final beneficiary. Values can be quantified in various ways, but provide a consistent metric for the direct comparison of the costs and benefits of alternative policy scenarios. Valuation is inferred by stakeholders and it is the engagement of these same stakeholders that is an essential component of the ecosystem services framework in that it helps to ensure that all views are considered when planning future options for the sustainable management of ecosystems. In fact, the scale at which humans as organisms perceive landscapes, that is, the "perceptible realm", is particularly important because this is the scale at which humans intentionally change landscapes, and these changes affect environmental processes and, thus, the delivery of ecosystem services.

At present, a number of legislation/policy targets include the requirement that ecosystem services and the ecological components underpinning them are considered. These include targets of "halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, including restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss". A second target, for 2050, states that: "By 2050, European Union biodiversity and the ecosystem services it provides, i.e. its natural capital, are protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human well-being and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided." While these targets reflect admirable attempts to address the inadequate consideration of ecosystem services in policy and management, much improved scientifically robust knowledge of ecosystems, the services they provide and the processes underpinning their sustainability will be required in order to meet these targets. Therefore, ecosystem services have been included as a core concept in Action 5 of the EU Biodiversity Strategy to 2020. This requires "Member States to map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020".

In conclusion, the incorporation of the ecosystem services framework into policy- and decision making has the potential to inform the effective and sustainable management of our natural resources, even though the knowledge gaps relating to the links between biodiversity, ecosystem function and human well-being must still be addressed. Nevertheless, the ecosystem services framework can be used as a means to better communicate and take account of nature's importance in policy- and decision making, with particular emphasis on human well-being, and on the conservation of the natural environment for reasons of inter- and intragenerational equity and bequest. However, challenges remain with respect to the effective communication of the framework to stakeholders at all levels.

1 Introduction

1.1 The Ecosystem Service Concept

The interactions between the Earth's physical and biological elements, which constitute the planet's ecosystems, are billions of years old and support all forms of life including humans. The Millennium Ecosystem Assessment (MEA, 2005) defines ecosystems as "a dynamic complex of plant, animal and microorganism communities, and the non-living environment interacting as a functional unit". Although both human survival and well-being are dependent upon global ecosystems, anthropogenic activities have degraded many of these ecosystems and this has had consequences with regard to what we derive from them (e.g. Daily, 1997; MEA, 2005; Lewis and Maslin, 2015).

For several millennia, human societies have recognised that benefits are derived, both directly and indirectly, from the environment and nature (Lele *et al*., 2013). The more recent concept of human benefits from the environment and nature emerged in the 1970s, initially described as "environmental services" (Wilson and Matthews, 1970). Later, Westman (1977) discussed the value of the benefits that ecosystems provide to human society, which he termed "nature's services". He further explained that the effects of the development and physical change imposed by human beings on ecosystems could potentially be quantified in order to inform society and, thus, influence policy and management decisions in order to mitigate ecosystem degradation. This concept became known as "ecosystem services" in the early to mid-1980s as ideas and understanding evolved (e.g. Ehrlich and Ehrlich 1981; Ehrlich and Mooney 1983); in the mid- to late 1990s, it slowly emerged as a potential framework for evaluating and protecting ecosystems and their biodiversity (e.g. Costanza *et al*., 1997). This, in turn, has resulted in the integration of the formerly separate disciplines of economics and ecology, and allows the investigation of the relationships between economies and natural environments (Costanza, 1991; Braat and de Groot, 2012).

In 2005, the MEA (2005) highlighted the fact that the previous 50 years had seen the most rapid loss and degradation of the Earth's natural capital and "ecosystem services", driven by the unprecedented rise in human populations and increasing per capita consumption. The report proposed the ecosystem services concept as a policy tool to achieve the sustainable use of natural resources in the future (Seppelt *et al*., 2012). During the same period, the Convention on Biological Diversity (CBD, 2000) adopted and endorsed the "ecosystem approach", defined as "a strategy for the integrated management of land, water, and living resources that promotes conservation and sustainable use in an equitable way", as a way of intervening to manage ecosystems, based on a systemic and participatory approach. The CBD later highlighted the benefits of using the ecosystem services concept to promote the ecosystem approach (CBD, 2006). Consequently, the concept of ecosystem services was included in the 12 principles of the ecosystem approach (CBD, 2004; see section 1.1.1) as a means not only of promoting the ecosystem approach, but also of embedding ecological thinking into policy and practice (Haines-Young and Potchin, 2009). More recently, the 2011 CBD Aichi Biodiversity Targets, Strategic Goal D and especially, Target 14 [https://www.cbd.int/sp/targets/ (accessed 14 June 2016)], called for the safeguarding of essential ecosystem services, especially those related to water and those that contribute to human health, livelihoods and well-being. This led to the inclusion of ecosystem services in initiatives such as Mapping and Assessment of Ecosystems and their Services (MAES), which is discussed further in section 4.1, and international and national biodiversity strategies (see Chapter 8).

The 2005 MEA report (MEA, 2005) and the evolution of the CBD led to the development of the "ecosystem services framework", which is also referred to as the "ecosystem services approach" (however, several authors, such as Tallis *et al*. (2008), Turner and Daily (2008) and Waylen *et al*. (2014), consider the term "ecosystem services framework" to be preferable to "ecosystem services approach" because the former is less similar to the "ecosystem approach" and, therefore, helps to reduce confusion). This ecosystem services framework is considered an effective means of communicating to all stakeholders, including the general public, the value of protecting ecosystems in order to maintain the flow of benefits (i.e. goods and services) that they provide for humans and their well-being. Theoretical estimates of this value can be used to provide a

rationale for extending conservation efforts with regard to land management through the design and provision of incentives and policies that are proportionate to the contribution of the services provided by an ecosystem (Hauck *et al*., 2013). However, in terms of ecosystem management, the "ecosystem approach", as endorsed by the CBD (see CBD, 2000), is, potentially, profoundly different from the "ecosystem services framework". These differences are outlined in section 1.1.1.

1.1.1 The ecosystem approach and ecosystem services framework

Both the "ecosystem approach" and the "ecosystem services framework" attempt to highlight the "ecosystem" as the basis for conservation, decision-making and policymaking; unfortunately, however, although both terms have often been used interchangeably (Waylen *et al*., 2014), they differ in their emphasis. The ecosystem approach focuses on natural processes and systems (e.g. Waylen *et al*., 2014; Martin-Ortega *et al*., 2015) and, consequently, is the basis for modern nature conservation in many parts of the world. In contrast, the ecosystem services framework (hereafter used throughout this review) focuses on understanding how natural systems and the linkages between ecosystem structures, processes and functions that lead directly or indirectly to valued human welfare benefits (Turner and Daily, 2008; Waylen *et al*., 2014). In the simplest of terms, the ecosystem services framework provides a way of understanding how nature delivers benefits and services for human well-being (Waylen *et al*., 2014). It also allows the evaluation of changes in specific ecosystem service flows and the comparison of previously incomparable resources (Toman, 1998; Salles, 2011); it may be perceived to be particularly powerful as a support tool for environmental concerns (Costanza *et al*., 1997). Although understanding a system in terms of flows of services can support holistic and equitable management, describing them in terms of ecosystem services is not the basis of the ecosystem approach (Fish, 2011). The ecosystem approach has influenced the ecosystem services framework, but ultimately the two approaches are different in their outlook and goals (Weylen *et al*., 2014; Martin-Ortega *et al*., 2015). A comparison of the two approaches or frameworks is shown in Table 1.1. For more information, see the review by Waylen *et al*. (2014).

1.1.2 Ecosystem services framework and its place within integrated water resource management and catchment services

Integrated water resource management (IWRM) is an umbrella term and is generally applied to the river basin or catchment (and sometimes country level); it promotes the co-ordination, development and management of water, land and related resources, in order to maximise economic and social welfare (GWP, 2000). Similar to this is the water–energy–food nexus [http://www.water-energy-food.org/ (accessed 14 June 2016)] which enhances water, energy and food security by increasing co-operation and efficiency, reducing trade-offs, building synergies, improving governance across

Table 1.1. A comparison between the ecosystem approach and the ecosystem services framework[a]

Ecosystem services framework[b]	Ecosystem approach
A framework for understanding how the biological components of an ecosystem (i.e. nature) deliver benefits and services for human well-being	A way of intervening to manage ecosystems using a systemic and participatory approach
Stakeholders are users or beneficiaries of ecosystem services. They provide knowledge and indicate interests that lead to ecosystem management and planning	Stakeholders provide knowledge and indicate interests that lead to decentralisation of ecosystem management and planning
Part of the process involves identifying which ecosystem services can be provided, how they can be provided and where they can be provided, with the goal of maintaining and improving human well-being	The process involves understanding the complex relationships that comprise socio-ecological systems, including the relationships between people and nature
Ecological functions and processes are included in supporting processes (also called supporting services)	Ecological processes and limits should be appreciated by all who contribute to decision making
The scale of influence is not explicit, ranging from local to global (see Table 5.1 for more details)	The scale of influence is not pre-set but decentralisation of management and planning is recommended

Table adapted from Waylen *et al*. (2014).

[a]Also see http://escom.scot/sites/default/files/resources/eco-communication2pager1.pdf (last accessed 14 June 2016).

[b]Also referred to as the "ecosystem services approach".

sectors and underpinning policy recommendations (Hoff, 2011; UN Water, 2014). The ecosystem services framework is very similar to IWRM and water–energy–food nexus (and ecosystem approach as highlighted above); they have common features in terms of their genesis, underpinning assumptions, and objectives and approaches, especially with regard to the sustainable use of natural capital and ecosystems (Niasse and Cherlet, 2015).

The proposal of "catchment services", which is based on the principle that an absence of joined-up thinking can lead to poor decision making, essentially promotes IWRM in which the catchment is the major unit of analysis (Daly, 2015a,b). Catchment services underlie the benefits (see Table 1.2) that the people living in and/or using the catchment (the river catchment is proposed as the land-based unit for water management and for most components of aquatic and terrestrial biodiversity management) receive, and they are derived from the components of natural capital, ecosystems, geosystems and human/social capital, present in the catchment (Figure 1.1; Daly, 2015b). However, the links between

Table 1.2. Description and examples of geosystem services and human/social system services

Geosystem services	Human/social system services[a]
The landscape geomorphology; bedrock and gravel; groundwater for drinking water and geothermal energy; soils and subsoils as chemical and physical attenuating media for pollutants; hydrometeorology (rainfall, evapotranspiration, wind); geological heritage sites; minerals; oil and gas; caves; cultural values associated with landscape features; etc.	Housing; farming, both intensive and extensive; mining; quarrying; wind farms; water abstraction facilities; roads; landfill sites; industry; cultural values associated with historical features and buildings, such as ring forts, castles and holy wells; water mills; pathways along streams and canals; other recreational facilities; etc.

[a]Human/social system services may be derived from various combinations of natural, financial, manufactured, human and social capital. Furthermore, financial, manufactured, human and social capital may be derived from outside the catchment boundaries. See IIRC (2013) and Forum for the Future (2015) for more details.

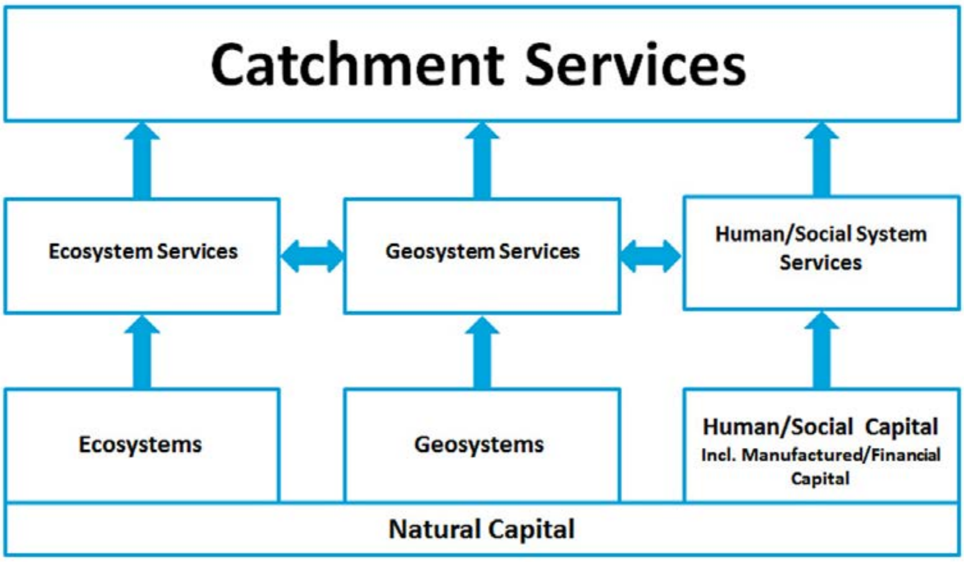

Figure 1.1. Schematic diagram of the different types of services provided to humans within catchments, as proposed by Daly (2015a), which encompasses services from the living, the non-living and the human/social elements, and how natural capital, consisting of ecosystem and geosystem services, links with human/social system services to give the holistic catchment services. Note: Human/social system services are typically derived from combinations of natural, financial, manufactured, intellectual, human and social capital. Furthermore, financial, manufactured, human and social capital may be derived from outside the catchment boundaries, and therefore this exact relationship may not hold true at catchment scale. See IIRC (2013) and Forum for the Future (2015) for more details. Figure reproduced courtesy of Donal Daly, Environmental Protection Agency.

natural (living and non-living) and human systems are complex, dynamic and multifaceted (e.g. Patterson *et al.*, 2013); nonetheless, Daly (2015a) argues that catchment services, as a framework for management, incorporate the perspective of local communities and stakeholders, being both comprehensive and inclusive of the complete mosaic of "physical, ecological, cultural and infrastructural features and functions within a catchment, thereby giving a sense of comfort that no one area is dominating and that the needs of local communities are taken into account". Regardless, ecosystem services, as one of the three main pillars (i.e. living components) of catchment services, are essential in the overall management of freshwater catchments. Furthermore, the use of the ecosystem services framework is an important means of ensuring that the role of ecosystems is prioritised in Water Framework Directive (WFD) goals and river basin/catchment planning, management and investment (Daly, 2015a).

1.1.3 The use of the ecosystem services framework

The application of the ecosystem services framework can maximise the benefits provided by ecosystems and enable improved environmental accounting and practice in the landscape (e.g. river catchments) (Ormerod, 2014; Daly, 2015b). Gordon *et al.* (2015) proposed that the ecosystem services framework is characterised by four core elements: (1) the recognition of the dependence of human well-being on the status of ecosystems; (2) an understanding of the biophysical processes and links that underpin the delivery of services; (3) the integration of natural and social sciences to better understand and communicate how services are delivered; and (4) the assessment of ecosystem services with regard to their incorporation into decision making. It is clear, therefore, that the ecosystem services concept and approach aim to provide an inclusive, integrated methodology for protecting the environment and its natural capital, while ensuring the sustainable flow of benefits for the human population.

1.2 Defining Ecosystem Services and Benefits

A plethora of terminologies relating to the ecosystem services concept has evolved in recent years and has, to some extent, complicated efforts to communicate the concept and its benefits to a wide range of stakeholders. All definitions vary slightly in wording but highlight the anthropocentric focus of the concept (Table 1.3).

Table 1.3. Examples of variations in the definition of ecosystem services and their source(s). This table is adapted from Nahlik *et al.* (2012) and developed further

Definition	Source
The benefits human populations derive, directly or indirectly, from ecosystem functions	Costanza *et al.*, 1997
The conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life	Daily, 1997
The capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly	de Groot *et al.*, 2002
The set of ecosystem functions that is useful to humans	Kremen, 2005
The benefits people obtain from ecosystems	MEA, 2005
Components of nature, directly enjoyed, consumed, or used to yield human well-being	Boyd and Banzhaf, 2007
The aspects of ecosystems utilised (actively or passively) to produce human well-being	Fisher *et al.*, 2009
A range of goods and services generated by ecosystems that are important for human well-being	Nelson *et al.*, 2009
Benefits that humans recognise as obtained from ecosystems that support, directly or indirectly, their survival and quality of life	Harrington *et al.*, 2010
A collective term for the goods and services produced by ecosystems that benefit humankind	Jenkins *et al.*, 2010
The direct and indirect contributions of ecosystems to human well-being	TEEB, 2010a
An activity or function of an ecosystem that provides benefit	Fisher and Turner, 2008; UK NEA 2011a; Mace *et al.*, 2012
The contributions of ecosystems to benefits used in economic and other human activity	EC, 2013a
The contributions that ecosystems make to human well-being	Haines-Young and Potschin, 2013

For example, the MAES process emphasises that ecosystem services represent “the realised flow of services for which there is demand” (EEA, 2015). Both Boyd and Banzhaf (2007) and Wallace (2007, 2008) describe ecosystem services as components of nature that are directly consumed, used or enjoyed by society and which cannot exist in isolation from the needs of society. The definitions most commonly adopted define ecosystem services as “the benefits people obtain from ecosystems” or “the contributions that ecosystems make to human well-being”, based on the MEA (2005) report or the Common International Classification of Ecosystem Services (CICES) report (Haines-Young and Potschin, 2013), respectively.

CICES (Haines-Young and Potschin, 2013) refers to “final” ecosystem services, which are distinct from “intermediate” services to avoid double counting in evaluation exercises. Services are described as “final” if they are the outputs of ecosystems (whether natural, semi-natural or highly modified) that most directly affect the well-being of people (see also Chapter 5 for more details). A fundamental characteristic of such services is that they retain a connection to the underlying ecosystem functions, processes and structures that generate them, that is, the so-called “intermediate services”. Boyd and Banzhaf (2007) also proposed that ecosystem services cannot include the ecosystem processes and functions that deliver the service, which they termed “intermediate products”. Thus, there is general consensus that there should be a distinction between “final” ecosystem services and supporting or “intermediate” services. This distinction is well illustrated in the ecosystem service cascade framework of Potschin and Haines-Young (2011) (Figure 1.2), which highlights the position of the CICES classification and the “production boundary” between social and economic systems and the environment. Therefore, as outlined in the COWI A/S (2014) report, supporting or intermediate services are “the biophysical/ecological structures, processes

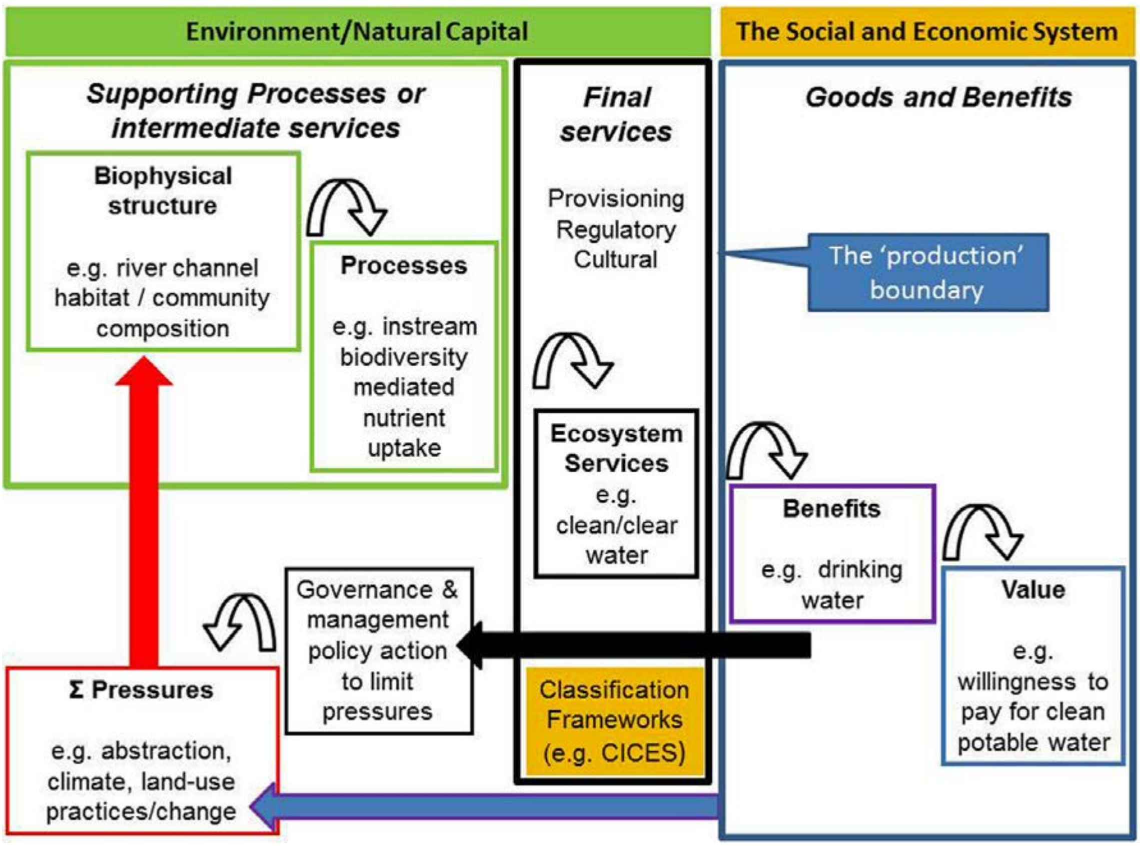

Figure 1.2. The ecosystem services cascade model highlights the role of supporting processes and intermediate services on the delivery of final services, and the goods and benefits that humans derive from the environment. Figure adapted from Potschin and Haines-Young (2011) (also see http://cices.eu/supporting-functions/).

and functions that underpin the provision of final ecosystem services" that humans benefit from either directly or indirectly (Costanza *et al*., 1997; Haines-Young and Potschin, 2010a,b, 2013; TEEB, 2010a; Mace *et al*., 2012). Furthermore, Fisher and Turner (2008) reiterated that the distinction between intermediate services and final services is not strict and depends on the beneficiary. It is also important to realise that a single ecosystem service (e.g. clean water) can be the product of two or more ecosystem functions, while a single ecosystem function may contribute to two or more ecosystem services (Costanza *et al*., 1997; Fisher *et al*., 2009; also see Figure 1.3).

1.3 Ecosystem Services Classification Frameworks

Classification frameworks are required to enable individual goods and services to be identified, quantified and evaluated (Haines-Young and Potschin, 2009; Landers and Nahlik, 2013). There have been several attempts to categorise and classify ecosystem services (e.g. de Groot *et al*., 2002; MEA, 2005; Wallace, 2007; TEEB, 2010a; Haines-Young and Potschin, 2013), but because of the range and complexity of ecosystem processes and the associated characteristics of ecosystem services, several different types of classification schemes have been proposed (Costanza, 2008). These include schemes that categorise ecosystem services by functional groupings, and examine the relationship between ecosystem processes and components, and services and goods, and the transitions from ecosystem processes and components to services and goods (de Groot *et al*., 2002); organisational groupings, such as services that are associated with certain species which regulate some exogenous input, or which are related to the organisation of biotic entities (Norberg, 1999); and descriptive groupings, such as renewable resource goods/non-renewable resource goods (e.g. Brown *et al*., 2007), physical structure services, biotic services, biogeochemical services, information services, and social and cultural services (Moberg and Folke, 1999).

The three key classification systems are the MEA (see MEA, 2005), The Economics of Ecosystems and Biodiversity (TEEB, 2010a) and CICES (Haines-Young and Potschin, 2013). The categorisation used by the MEA (2005) has been widely employed. Four main categories of ecosystem services are defined by the MEA: provisioning services; regulating services; cultural services, which are the so-called final ecosystem services; and supporting or intermediate services (see Table 1.4 for descriptions). A striking feature of the MEA (2005) categorisation, and, in some respects, other schema, is that certain services may fall within a number of groups, depending on the benefits to humans; for example, fish may be a "provisioning service" if they are a key food source or a "cultural service" if associated with recreation. TEEB (2010a) used similar provisioning, regulating and cultural groupings, but exchanged the

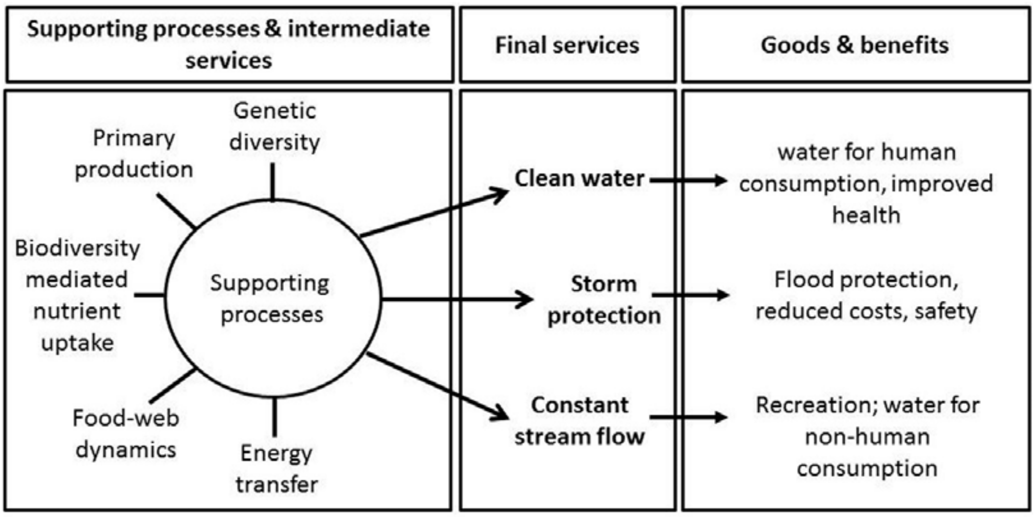

Figure 1.3. Conceptual relationships between supporting processes and intermediate services and final services for freshwaters, which, in turn, provide human society with goods and benefits. The figure also shows how joint goods and benefits can stem from individual final services. Figure adapted from Fisher *et al.* (2009).

concept of supporting services for a new category termed "habitat services", although this was essentially semantics. Three of these services (provisioning, regulating and cultural) are included in the CICES categorisation (Haines-Young and Potschin, 2013) (Tables 1.4 and 1.5).

The distinction between functions, services and benefits is important, especially for economic evaluations; however, it is often not possible to make a fully consistent classification, especially for regulating services (TEEB, 2010a). Nevertheless, in the CICES classification, a hierarchical structure has been employed so that the categories at each level are non-overlapping and without redundancy (Haines-Young and Potschin, 2013). CICES used the typology of ecosystem services suggested in the MEA (2005) as its starting point and

Table 1.4. Names and descriptions of the three main ecosystem services categories, with examples in the context of freshwater ecosystems[a]

Service	Description
Final services	
Provisioning	Readily understandable as the material or energy outputs from ecosystems, and include the supply of fish, food, fibre or other renewable materials (TEEB, 2010a; Bullock and O'Shea, 2013). For example, the supply of water for consumption, agriculture and industry are among the key provisioning services of freshwater ecosystems (Haines-Young and Potschin, 2013)
Regulating and maintenance	Sometimes referred to as maintenance services, these incorporate the various ways in which living organisms can mediate or moderate the ambient environment that affects humankind (Haines-Young and Potschin, 2013). For example, they ensure water quality by removing excess nutrients and degrading waste and toxic substances through living processes. Other examples include the regulation of local climates, water flow moderation and the regulation of human health (Haines-Young and Potschin, 2013)
Cultural[b]	These perhaps have the most variable definitions of all. They have been proposed to include the non-material benefits that people obtain from contact with ecosystems, such as direct or indirect benefits in the form of amenity and recreation and also certain non-use goods that are valued for their pure existence or which are perceived to contribute to quality of life (TEEB, 2010a; Bullock and O'Shea, 2013). CICES (Haines-Young and Potschin, 2013) categorised cultural services as the physical setting for recreational activity and for cultural values.
Intermediate services	
Supporting processes[c]	These underpin almost all other services. In freshwaters, they relate to all levels of aquatic biodiversity from genetic to community diversity, primary production and other ecosystem processes and functions that underpin well-functioning ecosystems and their resilience to internal and external pressures (Mace *et al*., 2012)

[a]Services are not fixed as "final" or "intermediate" and are dependent on the beneficiary. See Chapter 5 for more details.

[b]Fisher and Turner (2008) consider aesthetic and spiritual values and recreation to be benefits and not ecosystem services. This again highlights the need for an agreed language in order to improve communication with stakeholders. Kenter *et al.* (2014) further expanded the concept of cultural services to include "shared" and "plural" values that go beyond individuals' "self-regarding values" [see Kenter *et al.* (2015) for further discussion].

[c]Also known as "supporting services" and "habitat services".

Table 1.5. Some examples of the breakdown of ecosystem processes, the ecosystem services provided and the goods that can be valued for human well-being. See Table 1.6 for a more complex breakdown of ecosystem services and goods and benefits

Ecosystem processes ➡	Final ecosystem services ➡			Goods and benefits (valued)
Supporting processes/ services[a]	Provisioning services	Regulating services	Cultural services	
Primary production, nutrient cycling, food-web dynamics, biodiversity, energy transfer	Potable water, fish (as food), water for agriculture and industry	Water quality, flood regulation, flow regulation, decomposition, carbon, nitrogen, climate, pathogens, human health, waste disposal	Landscape aesthetics, spiritual/ religious/cultural, education, wildlife	Food, fibre, water for consumption, water for non-consumption, disease control, flood protection, tourism, cultural and historical heritage, recreation (hiking, angling, kayaking, etc.)

[a]Also referred to as "intermediate services".

refined it on the basis of many of the key criticisms discussed in the literature (e.g. Boyd and Branzhaf, 2007; Wallace, 2007, 2008; Costanza, 2008; Fisher *et al.*, 2009). At the highest level are three of the service categories used in the MEA: provisioning, regulating and maintenance, and cultural. Below these major "sections" are nested in a series "divisions", "groups" and "classes", as illustrated in Figure 1.4 (and also see Table 1.6 for more details and examples). The divisions give the main types of outputs or processes; these are then subdivided into biological, physical or cultural types or processes within the groups. At the class level, further divisions identify the specific outputs or processes.

The hierarchical structure of CICES handles many of the challenges that arise in relation to the different spatial and thematic scales used in different applications. Thus, as we move successively from sections, through divisions, groups and classes, the description of the service becomes progressively more specific and there may be many service types nested within these broader categories (Haines-Young and Potschin, 2013) (see Figure 1.4). The definitions of ecosystem services in CICES allow the user to consider the characteristics of the ecosystem service in question and, in turn, the natural capital, process and functions that produce them. Ultimately, by identifying the key characteristics it allows for better management, maintenance, restoration and overall evaluation of ecosystem services (Fisher *et al.*, 2009). Box 1.1 describes some recent developments in ecosystem services classifications in the USA. Examples of freshwater services within the CICES framework are provided in Table 1.6.

Box 1.1. Ecosystem service classification developments in the USA

Two ecosystem classification systems have emerged from the US Environmental Protection Agency (US EPA) after outside consultation: the Final Ecosystem Goods and Services Classification System (FEGS-CS) and the National Ecosystem Services Classification System (NESCS). Both emerged in parallel and focus on the "final ecosystem services approach". The final ecosystem goods and services (FEGS) concept is used to define, classify and measure ecosystem services (Landers and Nahlik, 2013). The focus of the FEGS-CS is on the beneficiary and the goods and services that require no labour or economic input for their generation. The system allows the separation of ecosystem services into intermediate services and FEGS. Two hierarchies are defined: environment and beneficiary. Two levels are defined for each hierarchy [i.e. three environmental classes (aquatic, terrestrial and atmospheric) with 15 sub-classes (rivers, lakes, etc.), and 10 beneficiary categories with 37 sub-categories (e.g. farmers, artists)]. Twenty-one categories of FEGS have been defined and, within these, a list of 338 actual FEGS have been defined. Sets of FEGS are linked to specific beneficiaries. Maps of "potential" FEGS can be produced using existing landscape classes. Links to two economic classification systems [the North

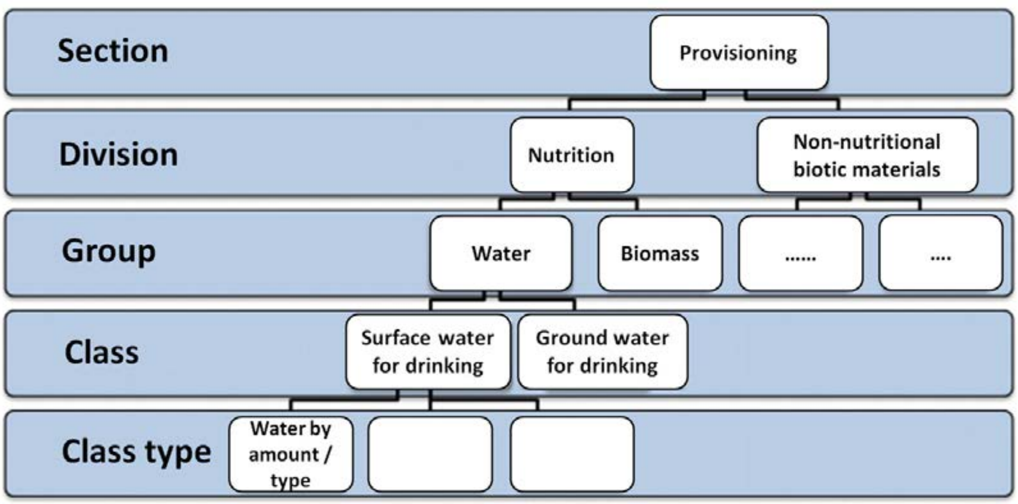

Figure 1.4. Example of the hierarchical structure of CICES (version 4.3). Figure adapted from Haines-Young and Potschin (2013). See also Table 1.6.

American Industry Classification System (NAICS) and the North American Product Classification System (NAPCS)] can also be illustrated.

The NESCS is based on the environmental classes and sub-classes from FEGS-CS and defines the components of nature that humans use (direct use) or appreciate (non-direct use). The final grouping ("direct user") lists the associated economic sectors that use the products of nature (i.e. the final ecosystem services). This classification framework is similar to the aforementioned NAICS and NAPCS. However, "NESCS is unique in its ability to identify both differences between users of the same final ES, and differences between uses for the same user, and is designed to meet the needs of policy makers" (C. Rhodes, US EPA, personal communication, August 2015).

1.4 Ecosystem Services Provided by Freshwater Ecosystems

Freshwaters, in their entirety, deliver an extensive range of ecosystem services (Maltby and Ormerod, 2011) (see Table 1.6). Provisioning services include water for drinking and non-drinking purposes (irrigation, cleaning, agricultural and industrial use), and the provision of food (e.g. fish). Regulating and maintenance services incorporate services that both directly (e.g. waste assimilation and pathogen control) and indirectly (e.g. regulation of decomposition, climate and flows) sustain environmental quality. Regulating and maintenance services ensure water quality and quantity by removing excess nutrients or moderating water flow, etc. (Lautenbach *et al.,* 2012). Climate regulation, through carbon and nitrogen cycling processes, maintains air quality, which influences the greenhouse effect and thereby regulates climate at both the local and global scales (de Groot *et al*., 2002).

Cultural services include tangible recreational uses (e.g. kayaking, angling and walking along a river) and also contribute to less tangible benefits, such as aesthetic or spiritual benefits as well as educational value. Tangible uses in Ireland, for example extensive recreational freshwater fisheries (game, pike and coarse angling), depend on less obvious aspects of ecosystems, such as good habitats and good visual appearance (TDI, 2013). However, it must be highlighted that not all freshwater habitats have the ability to support all processes and functions, or deliver services and goods equally (Maltby, 1986; Maltby *et al*., 1994; Bullock and Acreman, 2003). The delivery of freshwater ecosystem services and related goods is highly dependent on their location within the catchment and the service in question (Maltby and Ormerod, 2011). For example, the supply of water in Ireland at present is principally associated with rivers and lakes (71%) and, to a lesser extent, groundwater (29%) (Bullock and O'Shea, 2013). In contrast, wetlands play little or no role in the direct supply of water for consumption in Ireland, but do offer a wide range of services, including groundwater recharge for subsequent abstraction (Maltby and Ormerod, 2011). Regardless, the combination of river, lakes, wetlands and other freshwater ecosystems constitutes a large part of the natural infrastructure that supports human well-being and economic growth (e.g. Postel and Richter, 2003). However, it is the supporting services and their connection to and within the physical environment that underpin the provision of other freshwater goods and services. The next section outlines and discusses the role of biodiversity and healthy ecosystems in providing and sustaining ecosystem services.

Table 1.6. Ecosystem services provided by freshwater ecosystems, based on CICES

Section	Division	Group	Class	Examples
Provisioning	Nutrition	Biomass	Wild plants and animals and their outputs	Wild fruit/berries, game, freshwater fish (trout, eel, etc.); includes commercial and subsistence fishing and hunting for food
			Plants and animals from *in situ* aquaculture	*In situ* farming of freshwater fish (e.g. trout/salmon) and other products (e.g. cranberry growing)
		Water	Surface water for drinking	Collected precipitation, abstracted surface water from rivers, lakes and other open water bodies for drinking
			Groundwater for drinking	Freshwater abstracted from (non-fossil) groundwater layer or via groundwater desalination for drinking
	Material	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing	Fibres and other products, which are not further processed; chemicals extracted or synthesised from algae, plants and animals; includes consumptive ornamental uses. Peat/turf for energy use
			Materials from plants, algae and animals for agricultural use	Plant, algae and animal material for fodder and fertiliser in agriculture and aquaculture
			Genetic materials from all biota	Genetic material (i.e. DNA) from wild plants, algae and animals for biochemical, industrial and pharmaceutical processes (e.g. medicines, fermentation, detoxification); bio-prospecting activities (e.g. wild species used in breeding programmes, etc.)
		Water	Surface water for non-drinking purposes	Collected precipitation, abstracted surface water from rivers, lakes and other open water bodies for domestic use (washing, cleaning and other non-drinking use), irrigation, livestock consumption, industrial use (consumption and cooling), etc.
			Groundwater for non-drinking purposes	Freshwater abstracted from (non-fossil) groundwater layers or via groundwater desalination for domestic use (washing, cleaning and other non-drinking use), irrigation, livestock consumption, industrial use (consumption and cooling), etc.
Regulating and maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by microorganisms, algae, plants and animals	Biochemical detoxification/decomposition/mineralisation in freshwater systems including sediments; decomposition/detoxification of waste and toxic materials (e.g. wastewater cleaning)
			Filtration/sequestration/storage/accumulation by microorganisms, algae, plants and animals	Biological filtration/sequestration/storage/accumulation of pollutants in freshwater biota, adsorption and binding of heavy metals and organic compounds in biota
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems	Biophysicochemical filtration/sequestration/storage/accumulation of pollutants in freshwater ecosystems, including sediments; adsorption and binding of heavy metals and organic compounds in ecosystems
			Dilution by freshwater ecosystems	Biophysicochemical dilution of gases, fluids and solid waste, wastewater in lakes, rivers and sediments

Table 1.6. Continued

Section	Division	Group	Class	Examples
Regulating and maintenance	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates	Erosion/landslide/gravity flow protection; vegetation cover protecting/stabilising terrestrial ecosystems
			Buffering and attenuation of mass flows	Transport and storage of sediment by rivers and lakes
		Liquid flows	Hydrological cycle and water flow maintenance	Capacity of maintaining baseline flows for water supply and discharge (e.g. fostering groundwater); recharge by appropriate land coverage that captures effective rainfall; includes drought and water scarcity aspects
			Flood protection	Flood protection by appropriate land coverage
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance habitat and gene pool protection	Maintaining nursery populations and habitats	Habitats for plant and animal nursery and reproduction (e.g. microstructures of rivers, etc.)
		Pest and disease control	Pest control	Pest and disease control including invasive alien species
			Disease control	In cultivated and natural ecosystems and human populations
		Water conditions	Chemical condition of freshwaters	Maintenance/buffering of chemical composition of freshwater column and sediment to ensure favourable living conditions for biota e.g. by denitrification, re-mobilisation/re-mineralisation of phosphorous, etc.
		Atmospheric composition and climate regulation	Global climate regulation by reduction in greenhouse gas concentrations	Global climate regulation by greenhouse gas/carbon sequestration by water columns and sediments and their biota; transport of carbon into oceans (dissolved organic carbon), etc.
			Micro- and regional climate regulation	Modifying temperature, humidity, wind fields; maintenance of rural and urban climate and air quality and regional precipitation/temperature patterns
		Energy	Renewables	Various types of energy generated from use of natural water flows and oscillations, e.g. osmotic energy, hydropower
Cultural	Physical and intellectual interaction with biota, ecosystems and landscapes	Physical	Experiential use of plants, animals and landscapes in different environmental settings	*In situ* wildlife watching
			Physical and experiential interactions	Walking, hiking, boating and leisure fishing (angling)
		Intellectual and representative interactions	Scientific, educational, heritage, cultural, entertainment, aesthetic	Subject matter for research and education, both on location and via other media; historic records, cultural heritage (e.g. preserved in water bodies); *ex situ* viewing/ experience of natural world through different media; sense of place, artistic representations of nature
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes (environmental settings)	Spiritual and/or emblematic	Symbolic	Emblematic plants and animals
			Sacred and/or religious	Spiritual, ritual identity (e.g. holy places); sacred plants and animals and their parts
		Other cultural outputs	Existence	Enjoyment provided by wild species, wilderness, ecosystems
			Bequest	Willingness to preserve plants, animals, ecosystems for the experience and use of future generations; moral/ethical perspective or belief

2 The Ecosystem and the Provision of Services

A minimum level of ecosystem "infrastructure" is necessary for the delivery of the range of services required by humans (Costanza *et al*., 1997). The term "natural capital", coined in the 1970s, has re-emerged in recent years as a description of the Earth's stock of physical and biological resources [ecosystem capital resources (water, soil, air, etc.)]. The natural capital of an ecosystem provides a flow of ecosystem services, whereas the abiotic components provide depletable and non-depletable or renewable resources. There is general agreement that ecosystem or biotic services must be underpinned by ecological processes (e.g. Amigues and Chevassus-au-Louis, 2011) and should be distinguished from abiotic services, such as mineral extraction, power generation and navigation. These abiotic services were termed "environmental services" by the French National Agency for Water and Aquatic Environments (ONEMA) (Amigues and Chevassus-au-Louis, 2011), but are often referred to as "geosystem services", as previously mentioned (e.g. Daly, 2015a,b; see also Table 1.2). While there are benefits to separately considering the two groups of services, as outlined by COWI A/S (2014), there are instances in which the exploitation of an abiotic service may affect the biotic or ecosystem services. As highlighted in a recent discussion paper by Daly (2015a), not all of the physical elements of natural capital provide ecosystem services, but all elements of a catchment's natural physical capital (i.e. geosystem services) must be considered together with the human/social system (e.g. housing, land use and water abstraction) for effective management of water resources. The UK National Ecosystem Assessment (UK NEA) follow-on report (UK NEA, 2014) also presents a conceptual framework that includes governance structures and human and social capital for the delivery of the goods and services that contribute to human well-being.

The uptake of the ecosystem services framework requires a strengthening of our understanding of what underpins the delivery of ecosystem services and the communication of the importance of the linkages, highlighted in Figure 1.2, to policymakers and other stakeholders, especially those involved in the management of natural resources (TEEB, 2010b). Freshwaters are intrinsically linked to the landscape (e.g. Cummins, 1992; Alexander *et al*., 2007; Callanan *et al*., 2008) and ecosystem services derived from them are underpinned by processes that result from an interaction of the physical, chemical and biological components of water bodies and the associated drainage landscape (Durance *et al*., 2016). As precipitation falls across the landscape, run-off drains from the land forming streams, rivers, ponds, lakes, wetlands, groundwater and underground aquifers (e.g. Daily, 1997). This movement of water results in the key processes of physical and chemical erosion, nutrient transport and sediment formation. This, in turn, gives rise to freshwater habitats, food and energy, and ensures water quality through purification and dilution of pollutants. Furthermore, geochemical and biological processes within catchments provide a regulating service by mitigating or attenuating pollutants arising from anthropogenic sources (e.g. EPA, 2010a; Feeley *et al*., 2013). Similarly, chemical and biological processes within freshwaters can assimilate various pollutants (e.g. Balvanera *et al*., 2014), providing improved final services and resultant goods and benefits, such as water quality and healthy fish populations.

Freshwater habitats also have a role to play in flood protection, and in many cases flow provision. For example, aquatic habitats, such as wetlands and marshes, slow water flow and reduce flood risk (Zedler and Kercher, 2005). In turn, the floodwaters deposit supplies of sediment and organic matter that enhance the fertility of the surrounding landscape and also provide essential habitats for many invertebrate and vertebrate species, and increase the productivity of both the floodplain and the main river channel (Johnson *et al*., 1995; Daily, 1997; also see section 2.2 for disservices). Wetlands can also effectively remove nutrients, toxins (e.g. heavy metals) and sediments from through-flowing water, thereby improving water quality and regulating diseases (Zedler and Kercher, 2005). Similarly, undrained wetlands (e.g. peatlands) act as a carbon dioxide (CO_2) sink, reducing greenhouse gas emissions and, therefore, reducing the risks associated with climate change (Wilson *et al*., 2016).

Primary production bridges solar and biological energetics and influences ecosystems through the provision of the basal food resources on which the aquatic food-web

is established (Field *et al*., 1998; Jobbagy and Jackson, 2000). These biological communities also play a vital role in the regulation of water quality and disease control through filtering and the uptake of nutrients and pollutants. Other biological communities (e.g. bacteria, fungi and invertebrates) alter and transfer nutrients within an ecosystem through decomposition processes, as well as consumption and egestion of primary producers and detritus. The physical movement and disturbance of materials and sediments by biological communities via bioturbation (mixing and redistribution of sediments) and bioerosion [the biological breakdown of carbonate rock into smaller fragments by mechanical abrasion and chemical dissolution of calcium carbonate ($CaCO_3$); see Neumann, 1966] contribute to the provision and sustainability of ecosystem goods and services (Prather *et al*., 2013). This ecosystem engineering enables biological communities to physically change their environment through burrowing, feeding and locomotory behaviour which directly affects the sediment structure and composition in aquatic environments (Murray *et al*., 2002; Meysman *et al*., 2006; Prather *et al*., 2013). These dynamics facilitate the establishment and persistence of biological communities by creating habitats, hydraulic pathways, protection and refugia for large numbers of species. Similarly, these communities and structures, and the linkages they form, are essential in the maintenance of food-webs, which in turn support higher order fish, bird and mammal species across freshwater ecosystems (Mace *et al*., 2012; Prather *et al*., 2013). Ultimately, it is the natural range and diversity of species (i.e. biodiversity) that maintains the delivery of the extensive range of goods and services derived from freshwaters, as discussed in Chapter 4.

2.1 Service-Providing Areas

The spatial heterogeneity of ecosystems has been reviewed by several studies (e.g. Luck *et al*., 2003, 2009; Syrbe and Wahl, 2012; Burkhard *et al*., 2014). The spatial units that are the source of an ecosystem service within an ecosystem can be made up of both "hotspots" and "coldspots" of service delivery. Hotspots are areas or units of an ecosystem that provide large components of services in a comparably small area/spot (Egoh *et al*., 2008; García-Nieto *et al*., 2013), with coldspots being the opposite (Burkhard *et al*., 2014). Referred to as service-providing areas (Syrbe and Walz, 2012) or ecosystem service providing units (SPUs) (Luck *et al*., 2003, 2009; Burkhard *et al*., 2014), these areas include the sum of biodiversity and the associated traits (i.e. the biotic components of the ecosystem) required to deliver a given ecosystem service, as well as the physical or abiotic ecosystem components (Vandewalle *et al*., 2009; Syrbe and Walz, 2012). An example of an SPU, in the simplest of terms, is the "view" or visual area that can be seen from a particular location, providing an aesthetic cultural ecosystem service (Burkhard *et al*., 2014). The use of SPUs could be more appropriate than the use of administrative (or similar) units (Syrbe and Wahl, 2012). Freshwater SPUs can be made up of the entire catchment, but could also be located within the "wetted" area of the water body. The SPU approach is similar to the catchment services approach used in Ireland (Daly, 2015a, 2015b; see section 1.1.2).

2.2 Multiple Ecosystem Services, Trade-offs, Synergies and Disservices

Ecosystems provide multiple services and multiple goods and benefits to society. For example, a river system may provide several provisioning services, such as water for drinking, water for industrial use and fish as a source of food for local populations. These multiple services will have varying relationships with each other, some linear and some non-linear, and will interact on various levels (e.g. Bennett *et al*., 2009). These relationships, however, are not well understood (Tallis *et al*., 2008; Bennett *et al*., 2009), and therefore, ecosystem service trade-offs and synergies may arise (see Table 2.1) (e.g. Syrbe and Walz, 2012). Egoh *et al*. (2008) highlight how most ecosystem services are not good surrogates for one another and that the management of one ecosystem service will not necessarily also benefit other services, and may in fact reduce other services, goods and benefits. Bennett *et al*. (2009) stated that without knowledge about the relationships among ecosystem services, there is a risk of incurring unwanted trade-offs, which may result in dramatic and unexpected changes in the provision of ecosystem services (Table 2.1).

The management of ecosystem services and associated trade-offs (and synergies) can change the type, magnitude and relative mix of services provided by ecosystems, and in some instances a trade-off may be an explicit choice in the management of an ecosystem; however, in other instances, trade-offs may arise without premeditation and possibly even a lack

Table 2.1. Definition of ecosystem service trade-offs and synergies, with examples from freshwater ecosystems

Interaction	Definition	Example(s)
Ecosystem service trade-offs	These are situations in which one ecosystem service increases and another decreases. This may be due to negative interactions between services or different responses (i.e. one positive and one negative) to a particular driver/drivers or management option(s) (Bennett *et al*., 2009)	A trade-off may occur with water abstraction and the need for drinking water. Over abstraction will affect water levels and thus the water bodies' holding capacity for fish, which may be used as a food source or for angling. Similarly, managing a freshwater ecosystem for water abstraction (e.g. adding a weir or dam) may reduce suitable habitats for spawning fish, alter flow and flood regimes, and reduce angling potential
Ecosystem service synergies	These are situations in which two or more ecosystem services either increase or decrease. This may arise from interactions between ecosystem services or as a response to a particular driver/drivers or management option(s) (Bennett *et al*., 2009)	A synergistic relationship exists between catchment sedimentation and recreation opportunities in lakes and rivers. A reduction in sediment entering a water body will also increase the aesthetics and quality of that water body, thereby enhancing the opportunities for recreation

of awareness that they are taking place (Giller *et al*., 2004; Rodríguez *et al*., 2006; Bennett *et al*., 2009). Regardless, the analysis of trade-offs is one of the key steps in bringing basic science to policy application (see Chapter 8) in the ecosystem services paradigm (Mooney, 2009). The concept of trade-offs is very similar to that of ecosystem disservices. Lyytimäki and Sipilä (2009) described ecosystem disservices as "functions of ecosystems that are perceived as negative for human well-being". In freshwaters, disservices can result from natural phenomena, for example the damage caused by flooding or from anthropogenic activities, such as the side-effects of deliberate manipulation of a freshwater ecosystem (Lyytimäki and Sipilä, 2009). Sometimes, however, the relationship between ecosystem services and disservices and the relative magnitude of the effects are difficult to elucidate. For example, invasive freshwater zebra mussels can have a detrimental effect on infrastructure (e.g. piers, water treatment plants and boats) and fisheries but, at the same time, reduce nutrients and increase water clarity, which, in turn, may reduce the toxic effects on the ecosystem and improve the aesthetic appeal for recreation (Limburg *et al*., 2010).

3 Links with Biodiversity

Freshwater habitats cover less than 1% of the Earth's surface, yet support about 10% of all known species (Maltby and Ormerod, 2011). The MEA (2005) drew attention to the link between this biodiversity (e.g. species richness and functional diversity) and the maintenance of ecosystem services, and since then a considerable number of publications have highlighted the essential role played by biodiversity in the provision of ecosystem functions which deliver the goods and services we rely on (e.g. see Christie *et al*., 2006; Bullock *et al*., 2011; TEEB, 2010a; UK NEA, 2011a,b; Truchy *et al*., 2015). Biodiversity includes the vast array of species and their genetic variation, which represent the self-replicating building blocks of ecosystems; these interact and are inter-related with balance (how many of each element) and disparity (how different the elements are from one another), and, in turn, provide goods and services through ecosystem processes in a renewable manner (Stirling, 2007; Biggs *et al*., 2012; Mace *et al*., 2012). The role of biodiversity ranges from supporting ecosystem processes (e.g. primary productivity) to regulating services (e.g. water purification and disease regulation) to cultural services (e.g. education and a sense of place).

3.1 Quantifying the Contribution of Biodiversity to the Provision and Sustainability of Ecosystem Services

The specific details of the links between biodiversity and ecosystem services are largely unknown (Durance *et al*., 2016). Much of the evidence has come from studies of the relationship between biodiversity and ecosystem functioning (or processes). This evidence has been obtained both empirically (see review by Hooper *et al*., 2012) and through more theoretical studies (see review by Loreau, 2010, and Harrison *et al*., 2014), and has established that decreasing biodiversity (and its attributes) is, by extension, also likely to affect ecosystem services and vice versa (e.g. Woodward, 2009; Cardinale *et al*., 2012; Mace *et al*., 2012; Durance *et al*., 2016). These processes, along with the physical and chemical characteristics of ecosystems and the complex and diverse species interactions that occur within them, are the basis of ecosystem services, and the goods and benefits that society derives from ecosystems (Mace *et al*., 2012; Harrison *et al*., 2014). For example, it is known that communities of algae have relatively high levels of species richness, whose niche differentiation (i.e. the species' preferences for patches of different habitats/surfaces and flow velocities) contributes to their coexistence and leads to higher levels of nitrate uptake from the water than would be possible by any single species alone (Figure 3.1) (Balvanera *et al*., 2014); thus, higher biodiversity equals higher nitrate uptake, which in turn provides the final service of clean water. However, in this model, it is unknown whether or not species richness has effects on nutrient removal and storage. This "unknown" regarding nutrient removal and storage needs to be addressed if we are to arrive at a full understanding of the overall effects of species richness on water quality (Balvanera *et al*., 2014). Furthermore, much of the research suggests that the

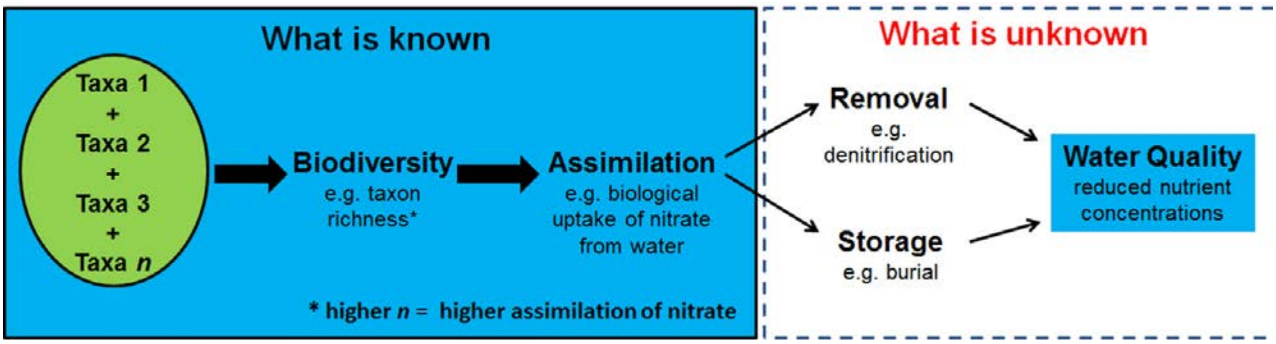

Figure 3.1. Understanding the role of biodiversity in ecosystem service delivery highlighted by the example of the known and unknown role of species richness on water quality. Figure adapted by the authors from Balvanera *et al*. (2014).

effect of biodiversity and ecosystem processes (e.g. nutrient uptake) may be non-additive, which generates uncertainty about the link between, and understanding of, biodiversity and delivery of ecosystem services (Truchy *et al*., 2015). However, some direct links between biodiversity and ecosystem services are more obvious. For example, Fahy (1995) found that anglers in Ireland reported that more coarse fish and, in general, greater diversity of fish species improved their experience (i.e. provided a cultural service).

Furthermore, confusion often arises because the single word "biodiversity" is used to describe a complex set of biological elements and concepts that help characterise the stocks and flows of biomass, energy and nutrients in ecosystems (Hairston, 1993; Mace *et al*., 2012). Evidence-based research highlights that the temporal and spatial organisation of "traits" within biological communities [e.g. the components of a species phenotype that regulate its responses to environmental factors, such as temperature, water chemistry, flow regimes and resource availability; see Truchy *et al*. (2015) for more details] are the driving forces of ecosystems and, therefore, affect the dynamics of populations, communities and ecosystem processes (e.g. nutrient cycling), which in turn give rise to ecosystem services (e.g. Kokko and López-Sepulcre, 2007; Bassar *et al*., 2010; Perkins *et al*., 2010, 2015; Truchy *et al*., 2015). Similarly, there is also still a general lack of understanding with regard to how the loss of particular species or groups of species will affect the sustainable delivery of ecosystem services (Balmford *et al*., 2011; Cardinale *et al*., 2012).

Harrison *et al*.'s (2014) review of biodiversity and ecosystem services highlights how research has begun to examine many of the relationships between ecosystem services and biodiversity, in order to determine the link between the biota or "providers" (along with their habitat requirements and land uses, etc.) and the provision and sustainability of ecosystem services. These "providers" are referred to as "ecosystem service providers" (ESPs) and are considered essential in the sustainable provision of a given ecosystem service (Kremen, 2005). ESPs can be defined as the "spatial units, component populations, communities, functional groups, interaction networks or habitat types that provide, support or are the source of the ecosystem service" [modified from the definitions provided by Kremen (2005), Zaccarelli *et al*. (2010) and Syrbe and Walz (2012)]. Therefore, ESPs are multifaceted and apply at various levels, for example on the population, functional-group and community scales (Harrison *et al*., 2014), but also on spatial scales (i.e. local, regional, global, etc.) (Zaccarelli *et al*., 2010). The rationale for ESPs is based on the concept proposed by Luck *et al*. (2003) of SPUs (see section 2.1). The combination of the two concepts can be referred to as the SPU–ESP continuum (Luck *et al*., 2009), which highlights the reliance of the biota on the abiotic environment.

3.1.1 Protected areas, biodiversity conservation and ecosystem service delivery

One of the driving principles behind the conservation of biodiversity is that it will ensure ecosystem functioning, which in turn secures the sustainable provision of ecosystem services, ultimately benefiting society (Naeem *et al*., 2012). The theory that biodiversity conservation aids the delivery of ecosystem services is not a new one (e.g. Eastwood *et al*., 2016), but, as mentioned previously, the links between biodiversity and ecosystem services are largely unclear (Durance *et al*., 2016; Eastwood *et al*., 2016). Nevertheless, the literature, although mixed, has highlighted some direct and indirect evidence to support conservation and improved ecosystem service delivery. For example, regulating and maintenance services and cultural services were shown to vary regionally with species of high conservation value and biodiversity in the UK (Anderson *et al*., 2008; Eigenbrod *et al*., 2009). On the European scale, Burkhard *et al*. (2012) found that important habitat classes for conservation (e.g. peat bogs and natural grasslands) ranked highly with regard to the supply of, relative to the demand for, regulating and maintenance services, but had a low ranking in terms of provisioning services. More recently, Castro *et al*. (2015) and Eastwood *et al*. (2016) found that protected areas can supply higher levels of some regulating and maintenance (e.g. climate regulation, erosion control, water flow maintenance and water quality) and cultural (e.g. aesthetics, education and tourism) services than non-protected areas, although their findings were not entirely conclusive. Eastwood *et al*. (2016) concluded that even though conservation areas provide higher levels of ecosystem services than non-conservation areas, it still remains unclear whether or not these higher levels of service delivery were a direct consequence of high levels of biodiversity.

Overall, the consensus in the literature is that biodiversity loss compromises ecosystem resilience (discussed in section 3.2) and, therefore, the capacity to sustainably deliver ecosystem services (e.g. Chapin *et al*., 2000, Diaz *et al*., 2005; UNEP, 2009; Mooney, 2009, as discussed in Chapter 4). However, much of the current research has provided insights into the difficulties and lack of current knowledge of the mechanisms linking various aspects of biodiversity to ecosystem functioning, and how the extrapolation of those results at larger scales and for the multiple processes contribute to the delivery of ecosystem services (Truchy *et al*., 2015; Durance *et al*., 2016).

3.2 The Concepts of Ecosystem Service Resilience and Thresholds

Fresh water is a finite resource. Consequently, any change to, or development of, freshwater resources will involve a trade-off between provisioning, regulating and maintenance, cultural and supporting services. Because provisioning services have a tangible or market value, whereas many of the other services fall within the domain of public goods, there has been a tendency to trade off (see section 2.2) these latter services in order to maximise the output of provisioning services. Increasing recognition of the consequences of such an approach has stimulated initiatives to address this issue and redress the balance (MEA, 2005; TEEB, 2010a; UK NEA, 2011b). Similarly, ecosystems and social systems are dynamic in their nature (e.g. Haslett *et al*., 2010) and, therefore, it is impossible to maintain social (see section 3.2.1) or ecological systems in a steady, unchanging state indefinitely, or to manage them for stability and security in a "command and control" fashion (Folke *et al*., 2010). Therefore, the concepts of ecosystem resilience and thresholds are critical to the maintenance of ecosystem services (Parsons *et al*., 2009).

Resilience (also termed "resistance") is the amount of change an ecosystem can undergo, or its ability to absorb disturbance and retain its essential structure and function (e.g. Walker and Salt, 2006). Ecological resilience considers how much change or disturbance an ecosystem can take before it is changed to another state. Thresholds mark the point at which the interplay between the pressure or stressor and ecosystem resistance is equal (Parsons *et al*., 2009) and can be defined as "boundaries in time and space in non-equilibrial systems that separate alternative stable states (i.e. dynamic regimes) organised around unique attractors or equilibrium points" (Briske *et al*., 2010). In other words, the addition and loss of species in a system are balanced until some stressor results in changes in speciation or extinction rates that change the system into an alternative stable state. The term "alternate state" generally describes the phenomenon whereby systems can exhibit a large change from one kind of regime to another (e.g. oligotrophic to eutrophic) (Walker and Meyers, 2004; Suding and Gross, 2006).

To date, only a few freshwater studies have evaluated the relationships among biodiversity, ecosystem processes and resilience, generally because of a lack of long-term data (e.g. Elmqvist *et al*., 2003; Woodward *et al*., 2015), while even less work has been carried out on ecosystem service resilience (Durance *et al*., 2016). Some evidence, mostly laboratory based, supports the "insurance hypothesis"; according to this hypothesis, maintaining levels of biodiversity will maintain the magnitude and stability of ecosystem processes and, in turn, maintain the services dependent on these processes, although not necessarily in the narrow sense of species richness (Durance *et al*., 2016). Additionally, organism identities, traits and body sizes are often at least as important, if not more so, than species richness (Perkins *et al*., 2010; Reiss *et al*., 2010): high levels of diversity in these dimensions imply enhanced resilience via complementary resource use or contrasting responses to disturbance events (Durance *et al*., 2016). Consequently, the identification and parameterisation of biodiversity is essential for the appraisal of disturbance events over relevant timescales, as they could conceivably range from intra-annual (e.g. for microbes) to inter-decadal (e.g. for invertebrates, fish and birds) (Durance *et al*., 2016), and, in turn, determine the resilience of associated ecosystem services.

The point (in space and time) at which a state changes is often referred to as the "tipping point". A change of state can alter the structure and effectiveness (either positively or negatively) of ecosystem functions (Jones and Sayer, 2003; Walker and Meyers, 2004), resulting in a possible change in the associated ecosystem services. The nature of thresholds and tipping points vary from system to system. These shifts, whether gradual or sudden, are generally thought to be driven by external perturbations, ranging from climatic fluctuations and overexploitation to eutrophication and invasive species (Walker and Meyers, 2004). In freshwaters, examples include the shift of a temperate lake with clear

oligotrophic waters to one with turbid eutrophic waters (Carpenter, 2003), a shallow lake with benthic vegetation to one with dense algae (Blindow *et al*., 1993; Scheffer *et al*., 1993; Scheffer, 1997; Jackson 2003) or a shallow lake with, potentially, alternate or dynamic states between plant- and plankton-based communities across a nutrient gradient (Jones and Sayer, 2003). In the Irish context, nutrient loadings from diffuse and point sources are the external drivers that may cause a tipping point in freshwater systems, potentially giving rise to alternative stable states (i.e. oligotrophic to eutrophic states). For example, in Lough Sheelin, which borders the counties of Westmeath, Meath and Cavan in Ireland (O'Grady, 2009), alternative states arose from eutrophication and invasive species. In the first instance, eutrophication (due to an increase in phosphorus) changed the lake from a brown trout [*Salmo trutta* (Linnaeus, 1758)] dominated system to a system dominated by coarse fish [i.e. the invasive roach *Rutilus rutilus* (Linnaeus, 1758)] (Figure 3.2). Then in the early 1990s, the reduction in phosphorus tipped the system back to a more balanced brown trout system (Figure 3.2). However, in 1998 the lake system changed state once again with a renewed increase in phosphorus, tipping the system back to a roach-dominated system (Figure 3.2). Finally, the introduction of the filter-feeding, invasive zebra mussel [*Dreissena polymorpha* (Pallas, 1771)] reduced the algal bloom and tipped the system back to a balanced system for the third time in 30 years (O'Grady, 2009) (Figure 3.2).

With regard to our understanding of tipping points and multiple states, what is unknown is the extent to which (singular or multiple) ecosystem services delivery is altered, degraded or enhanced by a shift in stable states. However, in the example highlighted above from Lough Sheelin, it is clear that when one service is degraded (e.g. brown trout angling), another is often improved (coarse fish angling) and vice versa (e.g. with the introduction of zebra mussels). Nevertheless, with regard to the understanding of the "cause-and-effect" relationship between drivers and pressures and ecosystems and ecosystem service provision, there is uncertainty in terms of how dynamic these systems are in the first place (e.g. Walker and Meyers, 2004). Nevertheless, Parsons *et al*. (2009) argued that knowledge on ecological thresholds should inform management decisions,

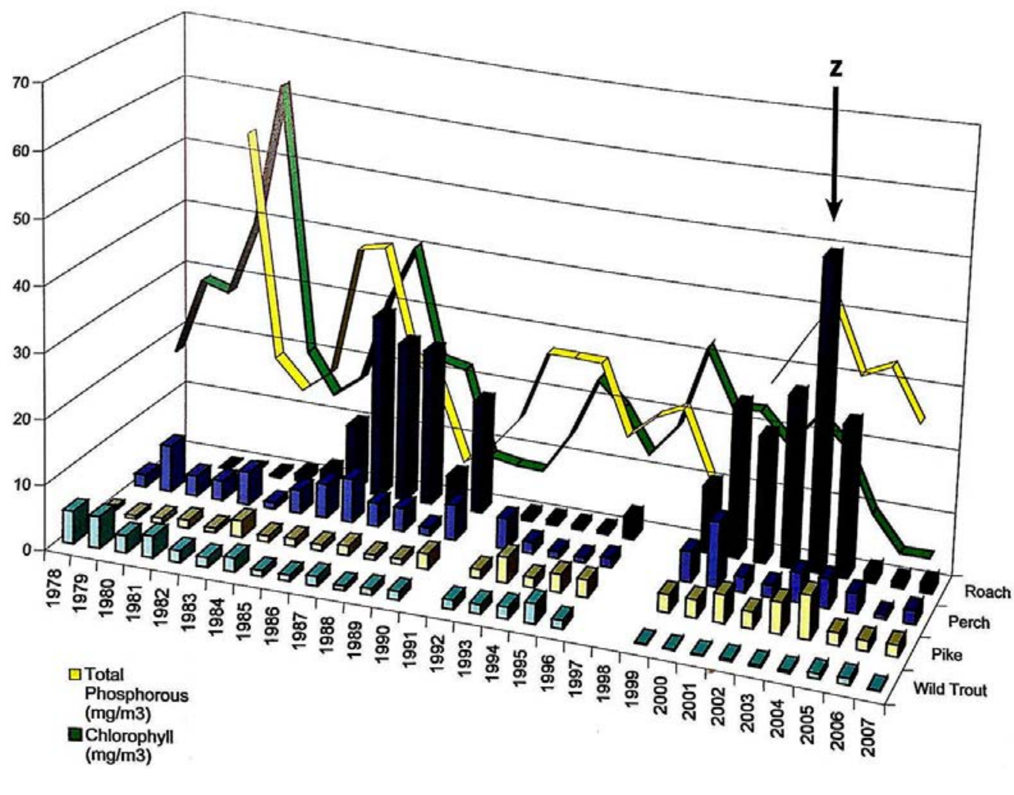

Figure 3.2. The annual chlorophyll, total phosphorus and catch per unit effort data for fish stocks in Lough Sheelin between 1978 and 2007. Invasive roach were first recorded in 1980 and zebra mussels (Z) were first observed in the lake in 2003. Figure reproduced from O'Grady (2009).

in order to prevent biodiversity and ecosystem service losses.

3.2.1 Linked social–ecological systems

Interestingly, Walker and Meyers (2004) highlighted that society and social systems can also change state, similar to that of ecological/environmental systems. Changes in, for example, social behaviour may lead to shifts in social structure, such as changes in population numbers, which, in turn, will change the demand for ecosystem services (e.g. clean water for abstraction). In so-called linked social–ecological systems (Walker and Meyers, 2004), changes in one system (i.e. the social system) can feed back as drivers and pressures to alter variables in the other system (i.e. the ecological system), and thereby cause a regime shift in that system. An example of a situation in which this may arise in Ireland relates to water demand and population growth in the Dublin city area, as highlighted by Kelly-Quinn *et al*. (2014). Interestingly, Walker and Meyers (2004) stated that the relationship between social and ecological systems may be unidirectional, resulting in a regime shift in only one system, but in some instances there may be a two-way interaction, resulting in regime shifts in both the ecosystem and society.

4 Information Requirements at Various Scales

The ecosystem services framework is an "inclusive, cross-sectoral decision-making process operating at appropriate spatial and temporal scales so that proper account is taken of the value of environmental systems for the well-being of people" (Haines-Young and Potschin, 2008). There are often many complex links in this chain, and many of these links have received little attention from the research community to date (Mooney, 2009). A conceptual step-wise approach was outlined by Mooney (2009; see Figure 4.1) and highlighted the long chain of knowledge required for incorporating ecosystem services into policy and management strategies. The chain starts at the nature of diversity and identifies the steps, including identification, mapping, assessment and valuation, needed to achieve the delivery of policy options for optimising ecosystem service delivery. Many of the steps require information and co-operation from the environmental and social sciences communities (Mooney, 2009).

4.1 Identification, Mapping and Assessment of Ecosystem Services

In Europe, MAES is an initiative of the European Commission to map and assess the state of ecosystems and their services, as required by Action 5 of the Biodiversity Strategy to 2020 (EC, 2011). The conceptual framework illustrating the link to ecosystem services is illustrated in Figure 4.2 and includes the following steps: (1) map ecosystems; (2) assess the condition of ecosystems; (3) assess the ecosystem services delivered by the ecosystems; and (4) perform an integrated ecosystem assessment. Within this common assessment framework, MAES provides a typology for ecosystems and ecosystem services (using CICES) and suggests indicators for the mapping and assessment of ecosystem condition and biodiversity. The key tasks involve the biophysical mapping and assessment of both ecosystems (12 types) and defined ecosystem services. The information will be used to assess the capacity of ecosystems to provide services at a European level and to integrate scenarios of future change and valuation into environmental and economic accounting (EEA, 2015). Ultimately, the mapping of ecosystem services is much more than just mapping the ecological functions that support the services; it also involves the identification of beneficiaries (see Chapter 5), their location and their use of the service or services of interest (Egoh *et al*., 2007), as well as informing management decisions and highlighting trade-offs (see section 2.2).

In Ireland, the initial mapping process commenced in July 2015. Led by the National Parks and Wildlife Service, it aims to further integrate environmental decision making and meet a number of targets under Ireland's National Biodiversity Plan by developing a national ecosystem methodology and an initial set of ecosystem service maps at various scales. It is anticipated that this will be the beginning of a national ecosystem assessment. The initial work has four main tasks: (1) to identify the most important ecosystem services in Ireland and understand which habitats or ecosystems support these services; (2) to develop indicators for selected ecosystem services, using existing and available data; (3) to collate and prepare data, and model and map the selected ecosystem services across Ireland; and finally (4) to present outputs and recommendations at a stakeholder workshop, outlining methods and identifying key gaps in knowledge and data. A select number of prioritised services will provide example outputs for the ecosystem service maps and these will help provide an

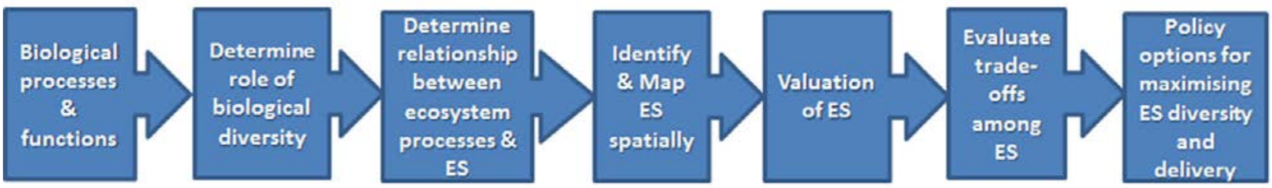

Figure 4.1. The chain of knowledge required to embed the ecosystem services framework in policy and management of freshwaters. Figure adapted from Mooney (2009). ES, ecosystem services.

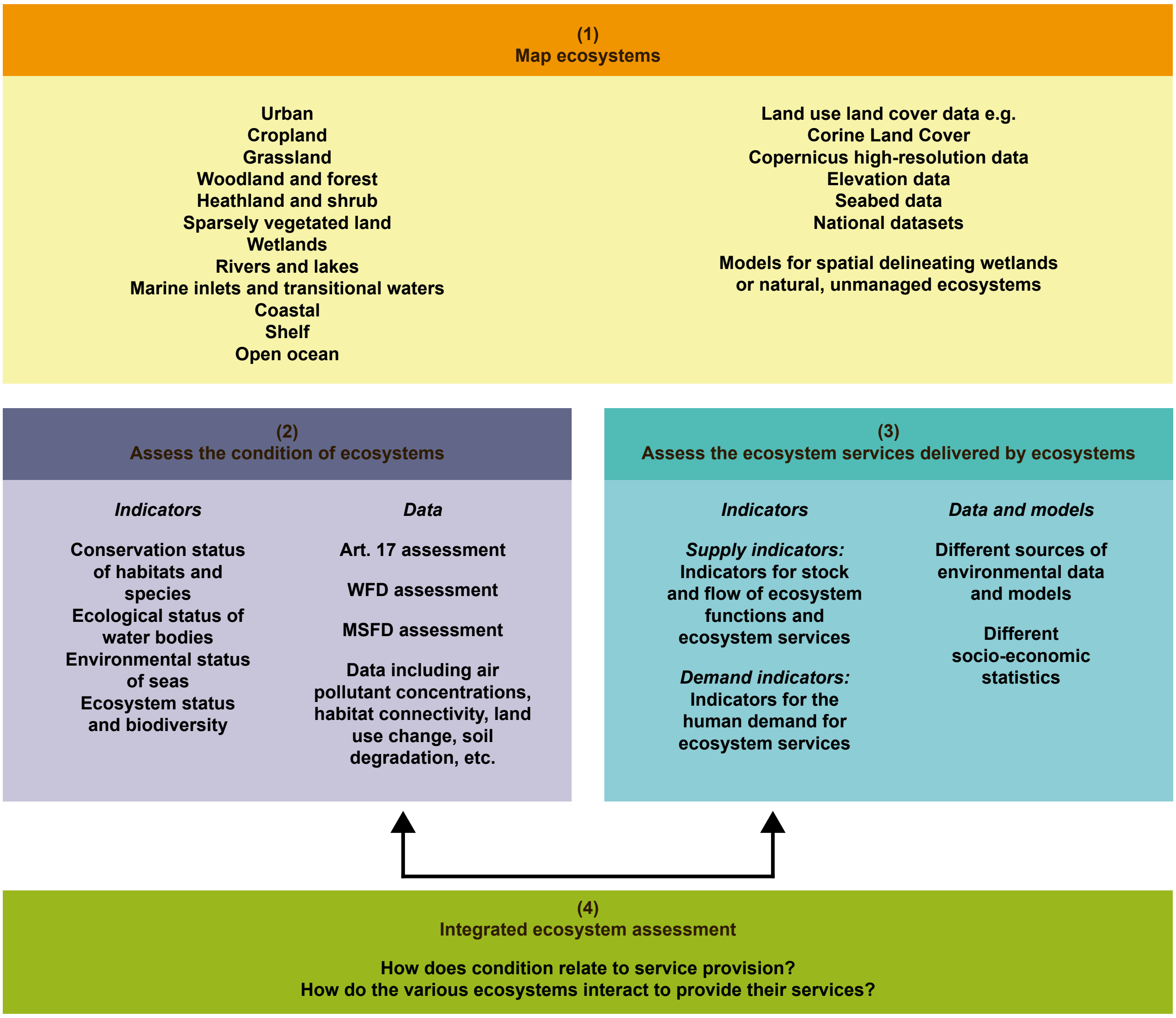

Figure 4.2. The common assessment framework for ecosystems and ecosystem services. MSFD, Marine Strategy Framework Directive. Figure reproduced from EC (2014) and EEA (2015).

evidence-based tool for policy development and land use management decision making, while the mapping exercise will provide the opportunity to identify data and knowledge gaps relating to individual services and will, therefore, help to target future research (Parker *et al*., 2015).

Because of ongoing societal development and needs, the influence and magnitude of negative drivers and pressures on ecosystem service delivery must be assessed (e.g. Metzger *et al*., 2006). Furthermore, global change gives rise to concern about alterations in ecosystem services, such as water supply, but the potential trajectories of change, especially on the regional scale, are poorly characterised (Schröter *et al*., 2005). Within the MAES approach, the DPSIR (Driving forces–Pressures–States–Impacts–Responses) theoretical framework is used to organise the data on the drivers (D) that generate the pressures (P) on ecosystems (S) and result in impacts (I), and which can be counteracted by relevant responses (R) (i.e. measures, management and policy). Some freshwater pressures identified by the European Environment Agency (EEA, 2015) are summarised in Table 4.1 and examples of human activity that impact the freshwater ecosystem and, consequently, the services, goods and benefits within them are highlighted in Table 4.2. Muller and Burkhard (2012) also support the use of the DPSIR framework for the assessment of ecosystem services and imposed it on the ecosystem service cascade (Figure 4.3) highlighted in Figure 1.2.

Table 4.1. Some examples of major pressures on freshwater ecosystems

Pressure	Cause
Habitat change	Modification of water course, flow regulation, fragmentation (dams)
Climate change	Changes in temperature and precipitation, changes in flow regimes, increased extreme events (e.g. droughts and floods)
Invasive species	Invasive alien fish, invertebrate and plant species
Land use/exploitation	Water extraction (surface and groundwater), gravel extraction, overfishing
Pollution and nutrient enrichment	Nutrient enrichment, acid rain, other pollutants

Source: Modified from EEA (2015).

Table 4.2. Examples of human activity on freshwater catchments and examples of ecosystem services and/or the goods and benefits at risk

Human activity	Impact on freshwater ecosystems	Ecosystem services/goods and benefits at risk
Wetland drainage	Eliminates key components and processes of wetland ecosystems, species habitats	Natural flood control and protection, habitats for fish and water birds and other biodiversity, recreation/tourism, natural water purification
Introduction of invasive species	Eliminates native species, alters production and nutrient cycling, reduces in-stream processes	Commercial fisheries and angling, affects a range of regulating and maintenance services (e.g. decomposition, water quality, native species habitat, disease control)
Land-use change/intensification	Alters natural chemistry of associated water bodies, run-off patterns, sediment dynamics, nutrient cycling, reduces in-stream processes	Water quality and quantity, species habitat, flood control, nutrient cycling
Emission of climate-altering pollutants	Alters natural precipitation patterns, run-off patterns, natural flow regimes, temperature	Water quantity, natural flood protection, species habitat, fisheries
Release of metals and acid-forming pollutants into the atmosphere or catchment	Alters natural chemistry of associated water bodies, nutrient cycling and in-stream processes. Eliminates naturally occurring species	Species habitat, fisheries survival and production, regulating and maintenance services (e.g. decomposition, recreation/tourism, water quality, human health)

4.2 Indicators Targeting Ecosystem Services

Assessing and quantifying ecosystem services is a challenging task, especially because the relationships among the goods and benefits, final services, ecosystem processes and functions, and the underlying biodiversity remain poorly understood (Kremen, 2005; Barbier, 2007; Hattam *et al.*, 2015). Nevertheless, the use of appropriate indicators can facilitate this process, providing information on thresholds, tipping points and overall changes in ecosystem service delivery (Hattam *et al.*, 2015). Indicators of sustainable, healthy, functioning ecosystems vary widely, but generally use a similar approach of comparing the ecosystems in question with identical (or as similar as possible) reference conditions, or through repeated ecological assessments (or monitoring), which then feed into management and policy plans (Rogers and Biggs, 1999). Therefore, indicators are proxies for complex phenomena and can be used to reflect the provision of a service and how it changes both spatially and temporally. In addition, indicators are useful, and often essential, for supporting management decisions and policies, as well as contributing to studies that aim to model and evaluate changes in ecosystem service provision (Niemeijer and de Groot, 2008; Hattam *et al.*, 2015).

Various spatially and temporally comparable indices have been developed to describe freshwater status and conservation status, and although not specific to ecosystem services assessment, they can nevertheless provide a measure of the ecological degradation of a water body, which is likely to reduce the provision and sustainability of ecosystem services (Suter, 2006; Bryan *et al.*, 2010). These include more traditional indices of abundance such as biomass (Tilman and Downing, 1994), evenness (Pielou, 1969), dominance (Simpson,

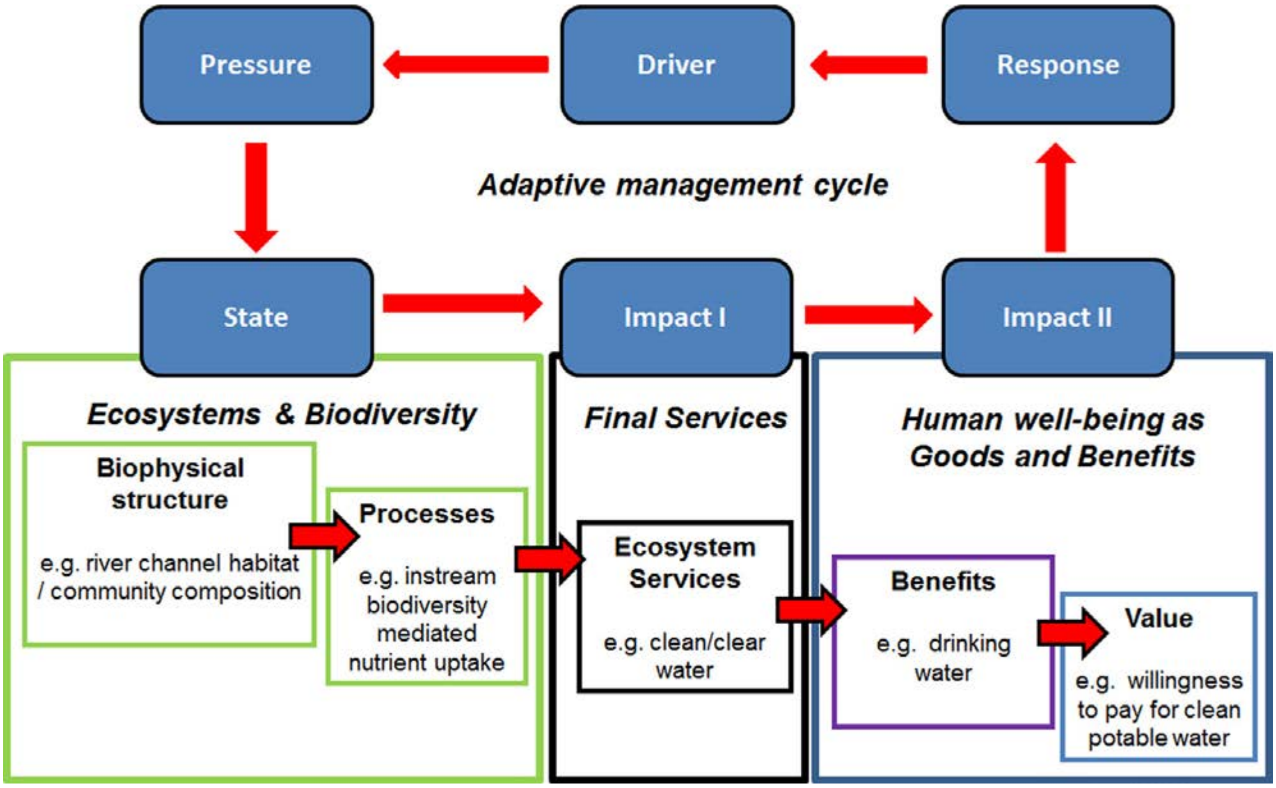

Figure 4.3. Ecosystem services as part of the DPSIR framework superimposed on the ecosystem cascade model outlined in Figure 1.2. Figure adapted from Müller and Burkhard (2012).

1949), rarity (Gaston, 1994), richness (Magurran, 2004) and diversity (Shannon and Weaver, 1949), but also WFD-associated biological metrics such as the Q-Value system (e.g. McGarrigle *et al.*, 2002), Small Stream Risk Score (SSRS; Kavanagh, 2006) or Acid Water Indicator Community species (AWICsp; e.g. Murphy *et al.*, 2013). These metrics, or a particular combination, could be adapted to target the management of ecosystem services based on social and economic values in the near future (Bryan *et al.*, 2010). Ringold *et al.* (2009) emphasised that ecosystem service indicators are not a substitute for ecological indicators, but instead complement them by communicating the role of ecosystems to policymakers and by facilitating valuation studies. Thus, indicators that measure supply and demand for the various services are required. No single indicator or group of indicators is likely to fulfil all the information requirements, and practical guidelines for selecting indicators relevant to targeting ecosystem services are still missing (Purvis and Hector, 2000; Bryan *et al.*, 2010; van Oudenhoven *et al.*, 2012). Ringold *et al.* (2009) proposed that indicators "should be strictly biophysical features, quantities or qualities that require little further translation to make clear their relevance to human well-being".

Although there is considerable discussion of indicators in the ecosystem services literature, relatively few indicators have been proposed or implemented to date. The Biodiversity Information System for Europe (BISE) [http://biodiversity.europa.eu/ (accessed 14 June 2016)] lists a limited number of available indicators for various ecosystem services. Those relating to freshwaters include the amount of water abstracted and aquaculture production, as indicators of provisioning services, and information on floodplain areas, records of annual floods and areas of wetlands in flood-risk zones, as indicators of flood protection. Recently, Maes *et al.* (2016) assessed the availability of indicators delivered by freshwaters across Europe. This study found that the best available information on indicators for provisioning ecosystem data was on freshwater abstraction and freshwater fisheries. Other provisioning indicators reported in the literature include "water crowding" (i.e. population supplied per unit accessible renewable supply; e.g. Rockström *et al.*, 2015) and use relative to accessible supply (i.e. the Water Poverty Index; e.g. Sullivan *et al.*, 2003). Indicators for regulating ecosystem services depend frequently on water quality monitoring reporting and the assumption that high water quality is positively related to the delivery

of ecosystem services. Most available indicators are for regulating and maintenance and provisioning services (JRC, 2012). Consequently, indicators of cultural ecosystem services are generally poor, despite the significant contribution of freshwaters to recreation (Maes *et al*., 2016).

A review by the Joint Research Centre (JRC, 2012) distinguished between "primary" indicators, which are proxies for the service being measured (e.g. "tourism attractiveness"), and "secondary" indicators, which are the underlying measures of the service (e.g. for tourism attractiveness, the secondary indicators could be "accessibility" and "naturalness"). The studies reviewed generally mapped freshwater provision and used secondary indicators, such as surface and groundwater availability. Similarly, Magnussen *et al*. (2015) listed examples of the direct and indirect (or secondary) indicators used to estimate the socio-economic value of provisioning and regulating and maintenance ecosystem services in Nordic freshwaters (Table 4.3).

In Ireland, the EPA have been examining the adoption of a "beneficial use index" in the context of scoring water bodies and prioritising the investment of resources for the successful implementation of the WFD. Under this approach, data relating to water quality or designated areas [Special Areas of Conservation (SACs) and Special Protection Areas (SPAs)] can be assembled into a spatial database and compared with data on population levels, abstraction needs and recreation activities. This beneficial use index could potentially be advanced to include data, for example from stated preference surveys, that can demonstrate the welfare value of rivers and lakes based on benefit transfer methods and distance decay factors, as demonstrated by several studies [see Hanley *et al*. (2003), Bateman *et al*. (2006) and EPA (2012a) for more details]. Wide-ranging and accurate data will be needed to include the value of provisioning services (e.g. water for human consumption), regulating and maintenance services (e.g. flood protection) and cultural services (e.g. access to areas of beauty), as well as the significance of non-use values (values not associated with use) (Bullock and O'Shea, 2013).

Table 4.3. List of identified direct and proxy indicators to estimate the socio-economic value of Nordic freshwater ecosystem services, potentially suitable for adoption at the national level

Ecosystem service	Identified direct indicator	Identified proxies
Fishing	(Market) value/value added of catch (sustainable) Number of jobs/employment/business/income	Size/value of catch (current amount or value) Number/percentage of fish and other species in commercial use
Fresh water (provisioning of) drinking and potable water, water for other human consumption	(Market) value/value added of (drinking) water, adjusted to reflect the real value (remove effects of any distorting subsidies)	Population/business served by renewable water sources
Water retention and purification and waste treatment	Value of regulating and protective function Replacement/avoided costs	Economic losses associated with lack of water quality (real or estimated) Population living in area depending (directly) on ecosystem based regulation
Natural hazards: mud flows/floods	Value of protective function Replacement/avoid costs	Economic losses associated with mud flow/flood (real or estimated) Population living in areas depending (directly) on ecosystem based regulation
Carbon sequestration and storage	Value of carbon sequestration and storage (e.g. based on CO_2 markets)	Costs related to climate change (real or estimated) based on, for example, climate-induced natural disasters

Note that the table does not address the issue of double counting, which needs to be considered when calculating aggregate values for multiple ecosystem services. Source: adapted from Magnussen *et al*. (2015).

5 Final Beneficiaries

The functions or processes of ecosystems become ecosystem services if there are beneficiaries (i.e. humans that benefit from them). In other words, the beneficiaries are what distinguish ecosystem services from ecosystem functions (Chan *et al*., 2006; Boyd and Banzhaf, 2007; Fisher *et al*., 2007). Therefore, beneficiaries are the interests of individuals and organisations (e.g. households, associations, societies and companies) that "drive active or passive consumption and/or appreciation of ecosystem services resulting in an impact (positive or negative) on their welfare" (Harrington *et al*., 2010; Nahlik *et al*., 2012; Landers and Nahlik, 2013). A knowledge of the location of the beneficiaries and the areas in which there is a demand for ecosystem services is essential for the design of environmental management policies (García-Nieto *et al*., 2013) and the appropriate targeting of management interventions (Chan *et al*., 2006). Similarly, the value, measurements and indicators used to assess ecosystem services will vary depending on the final beneficiary. For example, water regulation services are an intermediate input to the final service of clean water provision; in this case, the benefit is better water quality and the beneficiaries are society at large. The indicator could be, for example, treatment costs or human health. However, if the final service is fish production and the beneficiary is the angling community or fishmongers, then water provision would not be a final service but an intermediate one, while the indicator would be, for example, catch per unit area or fish size. Therefore, the identity of the beneficiaries, as well as what is being valued, measured and monitored, will determine whether the ecosystem service is considered final or intermediate (Boyd and Banzhaf, 2007; Egoh *et al*., 2007; Fisher *et al*., 2007). This may have implications for management and policy, especially with regard to multiple ecosystem services and trade-offs (see section 2.2). The role of beneficiaries as stakeholders is discussed further in Chapter 7.

5.1 The Scale of the Benefits: Temporal, Local and Global

Hein *et al*. (2006) stated that ecosystem services are supplied on various spatial and temporal scales (Table 5.1). The scales at which services are delivered and benefited from are often multifaceted; for example, climate and flood regulation can apply both locally and regionally, but also temporally across seasonal, annual and medium-term scales (e.g. Kremen, 2005; Burkhard *et al*., 2014). In freshwater systems, the range of spatial and temporal scales varies from the short-term, site level (e.g. water quality) to the long-term, global level (e.g. carbon sequestration) (Turner *et al*., 2000; Limburg *et al*., 2002). As a consequence, the stakeholders and beneficiaries of ecosystem services will also vary temporally and spatially, with the scale at which the ecosystem service is supplied determining which stakeholders may benefit (Vermeulen and Koziell, 2002; Burkhard *et al*., 2014).

Table 5.1. Temporal and spatial scales of the benefits, with some examples

Spatial		Temporal	
Local	Communities, rivers, lakes, wetlands, groundwater, ecosystems	Short term	Events, peak flows
Regional	Catchments, landscapes, administrative regions, river basin districts	Seasonal	Tourist seasons, growing seasons, peak flow periods
National/ international	Countries, international river basin districts	Annual	Yearly sums, average values and maximums
Continental	Europe, Asia, etc.	Medium term	Years, decades
Global	–	Long term	Multi-decadal, generations, centuries, millennia

6 Valuation of Ecosystem Services and Benefits

A key component of the ecosystem services framework is the economic valuation of changes in ecosystem services. The value can be quantified in various ways. Economic values use money as a measure. Money has the advantage of being readily meaningful to different stakeholders and policymakers. Consequently, economic valuation provides a consistent metric for directly comparing the costs and benefits of alternative policy scenarios (TEEB, 2010a). However, the use of monetary metrics is vulnerable to being incorrectly perceived as representing market values alone.

Economic markets, and market prices, exist for many provisioning services [e.g. commercial fish catches are estimated to be worth €0.75 billion to €0.83 billion to the Irish economy annually (TDI, 2013; NSAD, 2015)]. Even for provisioning services, values should represent those that best distinguish the value of the resource itself rather than supplementary human or capital inputs. These values should also be interpreted with care as they may fail to capture the value of the underlying stock of the resource and the sustainability of its harvest. Relevant markets are less common for regulating and cultural services. The absence of markets (and thus market prices) has led environmental economists to develop a suite of non-market valuation approaches that are capable of eliciting values for these services. The different approaches are described in more detail below.

6.1 Valuation of Freshwater Ecosystem Services and Benefits

The economic valuation of the benefits that people derive from ecosystem services may be undertaken by primary valuation research or value transfer (DEFRA, 2007). The former is always the preferred option whenever funds allow, as it provides the more accurate estimate on the basis of the local environmental conditions and the socio-economic context. Methods currently available for primary valuation, and comprehensively explained by Eftec (2006) and Christie *et al*. (2008), include:

- *Market prices*: used to assess the value of services that have direct market value. Predominantly, market prices are available for only provisioning services (e.g. the market price of fish).
- *Costs-based approaches* (replacements costs, damage costs avoided and production functions): these approaches assess value through the costs of providing the services elsewhere in the economy (e.g. the reduced costs of flood damage versus expenditures on the costs of an alternative flood management infrastructure).
- *Revealed preferences* [travel cost method (TCM) and hedonic pricing]: used if the values that people derive from freshwater services are revealed by their behaviour in related markets. For example, the TCM could use the cost involved (expenditures and time) in travelling to a river as a means to determine the recreational value of that river, while the hedonic pricing method uses evidence of house price premiums to measure how much people are willing to pay for the amenity benefits of properties close to, for example, a river.
- *Stated preferences* (contingent valuation and choice experiments): use surveys to ask people directly about their willingness to pay (WTP) (or willingness to accept compensation) for the services that are delivered by freshwater ecosystems. One advantage of the choice experiment method is that it allows the attributes of an environmental good to be valued.
- *Multi-criteria decision analysis* (MCDA): although not strictly a method of placing economic values on ecosystem services, MCDA provides a useful tool for directly comparing multiple criteria (such as ecological, economic and social criteria) in decision making, and is thus a useful tool for evaluating complex environmental issues (Mendoza and Prabhu, 2003). A recent development has been the adoption of the ecosystem services framework as the criteria in MCDA. For example, Zhu *et al*. (2015) used MCDA to monitor ecosystem service function in the Three River Headwaters Region of the Qinghai–Tibet Plateau, China, while Fontana *et al*. (2013) utilised stakeholder weights for ecosystem services to assess alternative landscape options.

The different valuation approaches have different merits, and no approach is universally suited to valuing the full range of ecosystem services (UK NEA, 2011a,b, 2014). The EU WFD "Aquamoney" project (Brouwer and Panagiotis, 2007) provides a useful summary of the applicability of the different valuation methods to the valuation of various freshwater ecosystem services (Table 6.1). Stated preference methods are among the most versatile, as they are capable of capturing a range of values, including non-use values (Mitchell and Carson, 1989), as well as providing the flexibility to assess the benefits of future scenarios, which may be beyond current levels of service provision. For example, the choice experiment approach requires respondents to complete a series of "choice tasks" that describe alternative future policy scenarios in terms of constituent attributes. An analysis of these choices enables values to be attained for a range of ecosystem service impacts of the policy.

Although choice experiments have been widely used to evaluate a range of environmental policies, the approach does require respondents to have a good understanding of the environmental goods under investigation. This is particularly challenging with regard to environmental goods that are complex in nature or of which people have only an indirect familiarity (Christie and Rayment, 2012). For instance, it can reasonably be argued that most people will have only a basic understanding of river systems and, therefore, may be unable to express their preferences for a particular river ecosystem.

Typically, environmental economists use interviews with stakeholders or focus groups to obtain an understanding of the extent of people's knowledge of environmental goods and how they are encountered, used or valued. Increasingly, economists are taking this process further by using one or more workshops as a means, not just of refining questionnaires, but of

Table 6.1. The applicability of valuation methods to freshwater ecosystem services, as indicated by Brouwer and Panagiotis (2007)

Water use being valued	Applicable valuation methods						
	Market analysis	Production function	Replacement cost/cost saving	Avoidance cost/averting behaviour	TCM	Hedonic pricing	Contingent valuation/ choice
Potable water for residential use	●	●	●	●		○	●
Water for irrigation	○	●	●				○
Water for livestock watering	○	●	●	○			
Water for food products and other manufacturing	●	●	●	○			
Cooling water for power plants	○	●	●	○			○
Commercial fishing	●	●	●	●			
Transport, treatment and medium for wastes	○	●	●	○			
Natural erosion, flood and storm protection		○	●	●		○	○
Sediment removal	○	●	●	●			
Biological diversity provision			●	○	●	○	●
Climate regulation (micro and macro)			●	●		●	●
Recreation (bathing, angling, etc.)	●	●	●	●	●	●	●
Cultural, historical and aesthetic values					●	●	●

Note: "●" represents methods that are very well suited to valuing the water services, while "○" represents methods that may be used to value the services.

providing people with sufficient understanding of the resource, or of the scenarios in question, to express reliable values in the course of the workshop itself. This process requires careful management to ensure that people are provided with a sufficient understanding of a scenario while, at the same time, not being encouraged to express values that are not representative of the true values they place on a resource. Christie *et al*. (2012) and others have administered choice experiments within such workshops to allow time for respondents to assimilate information and discuss and think about the environmental goods in the context of the range of environmental and non-environmental goods that they value. This approach has been shown to improve the accuracy of valuation. Approaches that are more discursive, deliberative and participatory can also be used to improve the accuracy of valuation surveys by allowing participants to not only gain a better understanding of the ecosystem resource, but also learn more about one another's perceptions and values (Whittington *et al*., 1992, 1997; Kenter *et al*., 2011).

An alternative approach to undertaking empirical valuation is to apply value transfer techniques that utilise existing case study valuation data to infer values for a new policy context (DEFRA, 2007). "Value transfer" saves time and money by reducing the need for new and resource-intensive empirical valuation work. It can provide reliable estimates of values in situations in which there is a high degree of similarity between an already valued site and the new study area or policy scenario, if, for example, it can be assumed that the benefit of a recreational visit to one river is similar to that of another. However, value transfer must be undertaken with caution given the likelihood of differences in the resource, or resource scenario, between different locations. In addition, Hynes *et al*. (2013) emphasised the importance of accounting for the cultural and behavioural differences of the people who are affected in the original case study site and those at the proposed policy site. More accurate value transfer methods make use of the original value function on which estimates were based and then calibrate this to account for differences in both the new location and the different population. In practice, this is difficult to achieve, particularly between countries, although value transfer can at least establish the direction and broad range of values.

Only recently has economic valuation been used to overtly address the issue of ecosystem services, rather than environment goods more generally (e.g. TEEB, 2010a; Bullock and O'Shea, 2013). Therefore, it will be important that any new empirical study of ecosystem services is designed with future value transfer in mind (Brander *et al*., 2006; Bateman *et al*., 2011). As discussed in section 6.2, there are currently very few valuation data relevant to Irish river ecosystem services and thus the adoption of value transfer will have limited success in the current policy context.

Typically, the values elicited using the methods outlined above relate to an individual's WTP for the ecosystem services in question. These values may then be multiplied by the affected population to give an aggregate value for the policy change (Klamer, 2003). Although such assessments of aggregate value are well recognised and are commonly used in cost–benefit analysis, more recent discourse has highlighted that such values may not capture the collective meanings and significance ascribed to natural environments. For example, an individual's preferences for the natural environment may be influenced by societal norms or an individual's behaviour might impact the welfare of others. Kenter *et al*. (2015) provided a comprehensive review of these shared and social values, and the methods that might be used to capture such values.

It is also evident that values may vary both spatially and temporally. For example, different ecosystem services will have different spatial impacts: the recreational benefits of angling may be limited to a particular stretch of river, while carbon sequestration may have global benefits. It will, therefore, be important to account for the spatial variation in the impacts of the values of ecosystem services when assessing the aggregate benefits. Furthermore, values are not stable and will vary over time. For example, both market and non-market values are influenced by a variety of factors (e.g. global markets, changes to environmental conditions and contemporary media stories). These factors, in turn, may affect market prices or change people's preferences. Although such factors are accounted for in economic theory, they do affect attempts to aggregate values that persist over longer timescales (e.g. carbon sequestration).

6.2 Case Study Evidence of the Value of Freshwater Ecosystem Services

Over the past few decades, there has been an exponential growth in the number of scientific papers that have evaluated the benefits derived from ecosystem

services. Many of these data have been captured by online data platforms, including the Environmental Valuation Reference Inventory (EVRI), Envalue, the Ecosystem Services Database (ESD) and the Review of Externality Data (RED). Although several thousand papers have been published (see Haines-Young and Potschin, 2010a), Seppelt *et al*. (2012) argued that the majority of these studies (1) are not based on primary data; (2) have inconsistencies in methodologies or (3) are not based on sound scientific evidence. Cowling *et al*. (2008) also noted that most assessments dealt with the biophysical elements, but gave little regard to social aspects. Indeed, when focusing on the value of freshwater ecosystem services, the TEEB (2010a) study identified only 10 value data points that it considered robust enough to be included in its assessment of the value of the ecosystem services provided by lakes and rivers.

In the following section, we provide a review of the evidence of the economic value of ecosystem services provided by freshwater systems. First of all, the data from Ireland are explored, then the scope is broadened to international studies. This evidence is also summarised in Appendix 1.

6.2.1 Irish studies on the value of freshwater ecosystem services

In Ireland, the studies that have aimed to evaluate freshwater ecosystems can be split into three groups: those that have assessed the condition of freshwater ecosystems; those that have assessed the value of freshwater ecosystem services to the general public; and those that assessed the value of, predominantly, water-based recreational services to specific user groups.

The condition of Irish rivers was assessed in 2000/2001 by M.C. O'Sullivan and Co. Ltd (MCOS, 2013), who developed a catchment-based water quality monitoring and management system for three Irish rivers: the rivers Boyne, Liffey and Suir. This study identified that the main stressors to river water quality were linked to agricultural activity, urban development and forestry. A more recent assessment of Irish water quality, undertaken for the EPA (2015), estimated that in the 2010–2012 period approximately 47% of the stream length and 57% of lakes did not reach "good" status, while only 1% of groundwater failed to meet "good" status. The failure of these freshwater systems to achieve good status was principally because of pollution from diffuse (54%) and point (46%) sources, generally related to phosphorus and nitrate loss due to human activities, such as agriculture and wastewater discharges to receiving waters from human settlements (EPA, 2010b, 2015).

There have been only a limited number of primary valuation studies that have assessed the economic value of water quality in Ireland. Doherty *et al*. (2014) conducted a choice experiment to evaluate public preferences for ecosystem services across different water bodies, including lakes and rivers. The attributes most highly valued in this study were "Water clarity and smell" (a change from "Poor" to "Good" or "Moderate" was valued at €42 or €30 per person per year, respectively) and "Aquatic ecosystem health: abundance and variety of fish, insects, plants, wildlife on the shoreline or banks" (a change from "Poor" to "Good" or "Moderate" was valued at €25 or €17 per person per year, respectively). Lower values were obtained for "Access to recreational activities", which included river bank activities and swimming. Stithou *et al*. (2012) also conducted a choice experiment study aimed at estimating the values attached to "good ecological status" in the Boyne River catchment. In this study, the highest values were found for "good" water clarity and appearance (€35 per person per year), "good" river life (€28 per person per year) and access to recreational activities (€23 per person per year). EPA (2014) utilised a contingent valuation study to assess Irish people's WTP to increase the number of river channels with a "good" status from the then current level of 68.9% to 100%. Mean WTP was €19 per respondent per annum, which they aggregated to a total of €65.35 million per annum. EPA (2012a) transferred value estimates from five European studies to rivers in each of Ireland's water management units. Bearing in mind the earlier discussion of the reliability of value transfer (see section 6.1), the EPA found that the corresponding estimates for the River Boyne were 76% higher than had been estimated by Stithou *et al*. (2012). Other studies that have used value transfer methods to evaluate Ireland's freshwater resources include those by DKM *et al*. (2004), CDM (2004) and Bullock and O'Shea (2013). The findings from all of these studies suggest that water quality is the service most highly valued by the general public, while recreational activities were also considered important.

Several studies have focused on the services valued by specific user groups, most notably kayakers and anglers. Hynes and Hanley (2006) used the TCM to address the trade-off between hydropower scheme

developments and white-water kayaking on the River Roughty. They identified that stressors to the natural environment include pollution, water abstraction from new housing, mining, forestry, hydroelectric schemes and pollution from farming. In a follow-up study by Hynes *et al*. (2007), they estimated that the welfare loss that would result if two rivers became unnavigable because of the building of a hydroelectric scheme would be between €2 and €36 per kayaker per trip on the Roughty river and between €10 and €55 for the River Boyne. They also found that, in their model, the value of water quality to kayakers was insignificant relative to kayaking-specific attributes associated with the rivers' white-water statuses. Curtis (2002) explored the impact of angling trips in Ireland. Mean travel cost per day was Irish £68 (or €86.34), of which £32 (€40.63) was for accommodation and meals, £14 (€17.78) accounted for fishing expenses and £22 (€27.93) was for actual travel expenses. Using the travel cost count model, Curtis (2002) then estimated that the consumer surplus value of an angling trip in Ireland was Irish £138 (€175.22) per day.

A socio-economic study was commissioned by Inland Fisheries Ireland in 2012 to assess the volume, value and economic impact of recreational angling in Ireland (TDI, 2013). The main findings of the study are as follows:

- Up to 406,000 individuals participated in recreational angling in Ireland in 2012.
- The total direct expenditure on recreational angling in 2012 was estimated to be in the order of €555 million, but this was estimated to have risen to over €600 million in 2015 (NSAD, 2015). Approximately 20% of this figure was generated by out-of-state anglers.
- The overall economic impact of recreational angling was estimated to be approximately €755 million (direct and indirect). Subsequent survey work and analysis in 2015 suggests that this figure has increased to €836 million (NSAD, 2015).
- Total tourist angling expenditure was estimated to be approximately €280 million.
- Recreational angling was estimated to support approximately 10,000 jobs in 2012 and this was revised to 11,000 jobs in 2015 (NSAD, 2015).
- Sea angling and freshwater Atlantic salmon and brown trout angling are the most popular categories for domestic anglers.

Respondents in the Survey of Recreational Anglers identified outstanding scenery and the friendliness/hospitality of the Irish people as the most appealing aspects of Ireland as a destination for recreational angling (TDI, 2013). A significant proportion of anglers also rated Irish angling highly in terms of its restful/relaxed ambience, the quality of the accommodation and the reputation of the angling "product". The improvement in water quality has been cited as the most positive development in angling in Ireland in recent years (TDI, 2013). However, there is evidence of a decline in recreational angling participation levels in recent years. Quality of angling, economic recession, poor weather, lack of fish, overfishing, illegal fishing, netting, commercial fishing, pollution and invasive species were all identified as factors that have contributed to dissatisfaction with current fish stocks.

The above review summarises evidence to suggest that the Irish public value rivers for water quality, ecological status and recreational opportunities, and the specific recreational user groups tend to place a higher value on their favoured recreational activity. However, there are likely to be other benefits associated with freshwater systems that have not been assessed in the Irish context, such as impacts on house prices, benefits to public health, visual impacts and impacts in urban areas.

6.2.2 International studies on the value of freshwater ecosystem services

International studies have evaluated freshwater ecosystem services over a range of scales from global and national assessments to individual river studies. The key findings of these studies are summarised below.

The TEEB (2010a) report provides estimates of the global ecosystem service benefits associated with the sustainable use of the Earth's major biomes. Value estimates were established using the value transfer approach and based on existing, robust value data: for the "lakes and rivers" biome, 12 value data points were used. Globally, TEEB (2010a) estimated that lakes and rivers have a value of between US $1779 and $13,488 per hectare per year, with (fresh) water supply (US $1141 to $5580), waste treatment/water purification (US $305 to $4978), and opportunities for recreation and tourism (US $305 to $2733) being the most valued services.

There have also been a number of national-scale ecosystem assessments including the UK NEA (2011a); the InDES project in India [http://www.indesprojects.com/ (accessed 1 July 2016)]; the river rehabilitation project in Israel (Garcia and Pargament, 2015); and an assessment of potable water supply in Malaysia (Othman *et al*., 2014). The UK NEA (2011a) is arguably the most comprehensive of these national assessments, with Chapter 9 (Maltby and Ormerod, 2011) of this assessment report focusing on freshwater ecosystems (open waters, wetlands and floodplains). The chapter by Maltby and Ormerod (2011) includes a detailed analysis of the ecosystem services provided by freshwater ecosystems, and an assessment of the condition, trends and drivers of changes affecting these ecosystems. Chapter 22 of the UK NEA also provides a review of the economic values provided by ecosystems. Key conclusions are that the water quality benefits of inland wetlands may be as high as £1500 million per annum, while planned river quality improvements may generate values of up to £1100 million per annum (UK NEA, 2011b).

The EU WFD (2000/60/EC) requires Member States to apply economic principles, methods and instruments to ensure “good ecological status” for all EU waters in the most cost-effective manner. The adoption of the WFD has led to an upsurge in studies evaluating European freshwater ecosystems. For example, in the UK, the NERA–Accent study (NERA and Accent, 2007) estimated household WTP for improvements in water quality, from “low” to “moderate” ecological status, to be £55 per annum. Morris and Camino (2011) drew on these estimates to assign values to improvements in water quality in both rivers and lakes. The UK Environment Agency subsequently used these values to estimate average benefits per kilometre of £15.60 for an improvement from low to medium, £18.60 for an improvement of medium to high, and £34.20 for an improvement from low to high. Hanley *et al*. (2006) administered a choice experiment to assess people’s preferences and the values of the likely key ecosystem service benefits provided by future improvements to two case study rivers: the River Wear (England) and the River Clyde (Scotland). The highest values were associated with maintaining bankside conditions (vegetation and erosion) and in-stream ecology (aquatic life: fish, plants, invertebrates), while aesthetic/appearance (amount of litter) had lower values for both rivers. Interestingly, values were significantly higher for the River Clyde; however, the authors found little evidence to support a value transfer between the two case study rivers. Poirier and Fleuret (2010) also used choice experiments to evaluate public preferences with regard to achieving good ecological conditions for four rivers in France. Values varied from €4.94 to €21.95 per household per year across the four rivers. Other studies include those by Polizzi *et al*. (2015), who combined stated preference with revealed preference methods to evaluate improvements in fish spawning conditions and recreational opportunities in Finland; Andreopoulos *et al*. (2015), who used choice experiments to evaluate people’s WTP for climate change adaptation policies that would improve the ecological status of the Aoos basin, Greece; Birol *et al*. (2008), who looked at different ways to reduce flooding resulting from intensive mining in the Upper Silesia region, Poland; and Johnstone and Markandya (2006), who carried out a study in England to address the recreational use of three river types, namely upland, lowland and chalk rivers. Using the TCM, Johnstone and Markandya (2006) found that high flow rates, biological quality and low nutrient pollution levels are the attributes that are most likely to affect the number of trips to a river.

The MARS (Managing Aquatic Ecosystems and Water Resources under Multiple Stress) project [http://www.mars-project.eu/ (accessed 1 June 2016)] is an EU-funded project that examines the impacts of multiple stressors (e.g. nutrient, sediment, temperature, etc., effects from urban and agricultural use, hydropower generation and climate change) on aquatic systems. These impacts are being investigated on three scales: the water-body, catchment and European scales. On each scale, the research aims to enhance the mechanistic understanding of how stressors interact and affect water resources, ecological status and ecosystem services, and to identify threshold responses to optimise reductions in stress (Hering *et al*., 2015). The project aims to adopt a conceptual model that will link the DPSIR risk assessment framework with the ecosystem services cascade model, thus bringing the ecosystem services paradigm into water resources management. Other relevant EU-funded projects include OpenNESS [http://www.openness-project.eu/ (accessed 1 June 2016)] and OPERAS [http://www.operas-project.eu/ (accessed 1 June 2016)], both of which are examining how ecosystem services might be best integrated into land, water and urban management policies and incentive mechanisms. The Natural Environment

Research Council (NERC) Valuing Nature Network project "BRIDGE: from values to decisions" [http://www.valuing-nature.net/values-decisions (accessed 1 June 2016)] also explored how ecosystem service values might be best embedded within decision making.

Two other significant and relevant projects are the DURESS (Diversity of Upland Rivers for Ecosystem Service Sustainability) project [http://www.nerc-duress.org (accessed 1 June 2016)] and the Wessex-BESS [http://www.brc.ac.uk/wessexbess/home (accessed 1 June 2016)] project. These projects are part of a major UK research council initiative, "Biodiversity & Ecosystem Service Sustainability" (BESS) [http://www.nerc-bess.net/ (accessed 1 June 2016)], which was designed to assess the role of biodiversity in delivering key ecosystem services in freshwaters. The DURESS project explores how future land use and climate change scenarios will affect upland river systems in Wales and, in particular, how river biodiversity mediates the delivery of river ecosystem services and the value of these services. Initial results suggest that water quality services tend to be more highly valued than in-stream biodiversity. Wessex-BESS investigates multiple ecosystem services delivered in the Wessex Chalk area as an example of a lowland, multi-functional, agricultural landscape, and has a sub-project on water-related services of fisheries and clean water. Two other projects, CBESS (Coastal Biodiversity and Ecosystem Service Sustainability) [http://synergy.st-andrews.ac.uk/cbess/ (accessed 1 June 2016)] and the F3UES project (Fragments, Functions, Flows and Urban Ecosystem Service – formally Urban BESS) [http://bess-urban.group.shef.ac.uk/ (accessed 1 June 2016)], are also included in the BESS programme and were designed to answer fundamental questions about the functional role of biodiversity in key ecosystem processes and the delivery of ecosystem processes on the landscape scale.

Outside Europe, the majority of freshwater valuation studies have been conducted in Canada, the USA and Australia. One of the earliest studies that used choice experiments is the study by Adamowicz *et al*. (1994). This study focused on assessing water-based recreational activities in Alberta, Canada. It measured the value to people of a wide range of benefits: landscape terrain, fish size, catch rate, water quality, facilities (e.g. campsites), swimming, beaches, distance from home, standing water, fish species, boating and running. The authors combined and compared the travel cost and choice experiment methods, which allowed in-depth individual and joint analyses. In Australia, Burton *et al*. (2007) used choice experiments to explore the ecological aspects of the Moor Catchment and its impact on farming. The three main environmental stresses identified were salinity, eutrophication and flooding. The impact on the area of farmland, the area of farmland planted with trees, the off-farm wetlands, the risk of major floods, the farm income and the associated value loss were also estimated. Heberling *et al*. (2000) also used choice experiments to assess the impacts of acid mine drainage in Pennsylvania, USA, on water quality and the accessibility of rivers for recreational use. Christie and Azevedo (2009) used both the choice experiment and contingent valuation methods to assess the benefits of lake water quality improvements at Clear Lake, Iowa, USA. Their results suggest that both local people and visitors valued improvements to water colour and clarity, the reduction of algal blooms and lake odour, and the increase in fish populations. Furthermore, their study provides evidence of convergence validity between the two valuation methods.

6.3 Methodological Issues with Regard to Valuing Freshwater Ecosystem Services

It is useful to highlight some methodological issues identified from these studies. First, it is clear from the literature that the use of focus groups has been essential for survey design (Bateman *et al*., 2006; Heberling *et al*., 2000; Hanley *et al*., 2006; Doherty *et al*., 2014). Engaging members of the public helps to ensure that the ecosystem services included in the studies, together with their representation, best reflect people's priorities (Bateman *et al*., 2003). However, it is also important to have input from stakeholders and experts (managing bodies and/or ecologists) to identify the nature of ecosystem functions and any threats to ecosystem services (Hanley *et al*., 2006; Doherty *et al*., 2014).

For stated preference studies, it is important to identify an appropriate payment vehicle (i.e. the means by which survey respondents can commit to paying for the environmental good). The aim of such studies is to identify the payment vehicle that maximises respondent buy-in and thus reduces protest bids whereby respondents decline to provide a WTP value. The most common payment vehicle used in the studies examined above was a fixed or percentage increase in the

water rates paid for a certain period, or an increase in national or local taxation, or fees from water operators or local authorities (Heberling *et al*., 2000; Bateman *et al*., 2006; Hanley *et al*., 2006; Birol *et al*., 2008; Stithou *et al*., 2012; Zhao *et al*., 2013; Doherty *et al*., 2014; Andreopoulos *et al*., 2015). Water charges have not been adopted as a payment vehicle in Irish studies, as these charges were only introduced for a short period before being suspended owing to political pressure and high levels of non-compliance; in general, they have been the source of much contention with regard to issues unrelated to water quality (Stithou *et al*., 2012).

Further issues relate to the difficulty of communicating ecosystem functioning to people, including aspects such as inter-relations between species, scientific uncertainties, the role of resilience and the existence of possible thresholds. Workshops can be used to enhance people's understanding of these issues, but are still restricted by the time available, the nature and representativeness of the audience, and the cognitive challenges related to people's response to hypothetical and real situations, including the role of uncertainty and risk.

Other issues relate to the use of utility-based valuations, particularly if monetary measures are used. There is a risk that such methods may fail to account for benefits that are realised at a societal or community level, or that they may not take into account ethical or more biocentric perspectives. Researchers have experimented with both monetary and non-monetary methods, discursive exercises and the elicitation of underlying socio-cultural values. For example, "citizen juries" – which are essentially focused workshops during which participants consider and debate a range of evidence from the perspective of *citizens* rather than *consumers* – have been used. No method that is satisfactory at all levels has yet been identified, particularly if quantitative estimates are sought. However, methods of communicating complex information and of eliciting more reliable and representative values are improving all the time; this will be a particular focus of the ESManage project.

6.4 Recommendations for Valuing the Ecosystem Service Benefits Derived from Freshwater Systems in Ireland

The literature on ecosystem services valuation reviewed above indicates that there have been a wide range of studies on the ecosystem service benefits of freshwater ecosystems. Studies have varied from global assessments to evaluations of individual rivers. Choice experiments have been the most commonly used approach to assign values to a range of ecosystem services; although the TCM has often been used to evaluate recreational benefits. The services most often investigated are river biodiversity, overall landscape condition (against flooding risk), water appearance (colour and smell) and recreational activities. Furthermore, the majority of the studies were administered to the general public, with a limited number of studies focusing on specific user groups, most notably anglers. Value transfer has also been applied if time and budget constraints restricted the collection of primary data. However, the wider applicability of the conclusions from these studies should be treated with caution because of the limited amount of value estimates available.

It is clear that there are significant challenges to undertaking ecosystem assessment and ecosystem service valuation. Specifically, a researcher must understand (1) the complex ecological linkages between biodiversity (the ecosystem) and ecosystem service provision; (2) how to convert this understanding into projections of changes in ecosystem service provisions that can be understood by the wider public; and (3) how to identify the means by which the public value the changes to ecosystem service provision. In this regard, there are many gaps in the current knowledge and high degrees of uncertainty are involved. Haines-Young *et al.* (2007) provided a useful framework for considering these complex linkages between biodiversity, services and benefits (Figure 1.2). Key to this framework is the need to present ecosystem services in terms of the "final products" that can be consumed/valued by people. Such an approach will make the valuation process more meaningful to the public solicited for the purpose of valuation and to policymakers and decision makers.

To be transparent and comprehensive, an ecosystem assessment needs to start from a position in which the full ranges of services are considered. The CICES (Haines-Young and Potschin, 2013) classification (adapted in Table 1.6 to focus on freshwater ecosystem services) provides a useful starting point for identifying the possible range of services. The assessment then requires the input of ecological knowledge to analyse whether or not, and to what extent, drivers of change will affect the delivery of these services. The next step is then to identify which of these services are likely to

impact on people's welfare (e.g. through the use of focus groups or deliberation) and to estimate an economic value for these key service impacts. Different valuation methods will be suited to the valuation of different services; however, the choice experiment method has been demonstrated in the literature to be the only method capable of valuing a wide range of services within a single survey instrument. The final stage in an ecosystem service assessment is to embed the results into river management and policy decisions. The use of the values of ecosystem services in policy decisions requires effective communication of the linkages illustrated in Figure 1.2 to policymakers and other stakeholders, especially those involved in the management of natural resources. Background data on the state of Ireland's freshwaters and the services delivered, and the extent to which the key stressors are likely to increase in the future, are presented in a report on Ireland's freshwaters [Work Package 2 (Feeley *et al*., 2016) – ESManage Project: www.ucd.ie/esmanage (accessed 1 June 2016)].

7 Stakeholders and the Participatory Approach

Stakeholder engagement is an essential component of the ecosystem services framework in that it helps to ensure that all views are considered when planning future options for the sustainable management of ecosystems. Indeed, in our review of valuation methods (see Chapter 6), it was noted that embedding participatory and deliberative approaches into the valuation framework may be a useful way in which to enhance respondent engagement with the valuation process.

It is useful to involve all relevant stakeholders in decisions that affect local ecosystems, particularly if effective management requires a change in human behaviour. Stakeholders include those whose actions affect the provision of ecosystem services and also those whose well-being is affected by an alteration in those services (Brauman *et al*., 2014). However, it can be difficult to devise systematic methods for incorporating stakeholder opinions and preferences, particularly in the context of remote rural environments. Villamor *et al*. (2014) reported on three methods used in a series of case studies and concluded that the methods adopted must be tailored to the particular context. Engaging stakeholders in a meaningful way requires careful thought and planning to ensure their active participation. This must be accompanied by appropriate methodologies and tools to deal with the complexity involved in managing a truly participatory process. Examples of methods include social network analysis (SNA), which has been used to involve local organisations in the overall goal of improving catchment water quality (Rathwell and Peterson, 2012; Stein *et al*., 2011; Hirschi *et al*., 2013; Hauck *et al*., 2015) and to connect health benefits to ecosystems (Berbés-Blázquez *et al*., 2014). Even in urban areas, where distances between communities are much shorter and physical isolation is of less obvious concern, SNA has been used to explain why agreement on a strategic vision for the environment of an urban river corridor can be difficult to achieve and often leads to some of the participants in the process feeling marginalised (Holt *et al*., 2012).

Difficulties may arise at all stages of the development of the adaptive management approach, including the early stages. For instance, presenting an already formed scientific consensus or draft policy to traditional stakeholder groups is not conducive to building trust and can reinforce existing positions or power relationships and frustrate efforts to negotiate compromises (Folke *et al*., 2005). This was evident in the implementation of the first cycle of the WFD in Ireland (Bruen *et al*., 2010): the Advisory Council mechanism relied on information provision and "consultation", but did not permit social learning to occur through the active participation of stakeholders. Other commentators go further and argue that a new approach is required that is better suited to dealing with the multiplicity of perspectives and uncertainties involved and facilitating a participatory approach (Cornell *et al*., 2013). This observation is supported by practical experience in many contexts. For example, a review of the approach to ecosystem services in Argentina found that there was an imbalance between the focus on generating general scientific knowledge and accessing local stakeholder knowledge, and that there was a deficit in the integration of both scientific and local stakeholder knowledge with participatory decision making (Mastrangelo *et al*., 2015). Some efforts to address these issues have been made in other disciplinary areas, for instance the production of community flood maps (Padawangi *et al*., 2016); in a system of payments for environmental (forestry) services (Le Coq *et al*., 2015); the mitigation of the effects of extreme rainfall on nature reserves (Tomczyk *et al*., 2016); and the identification of the potential for integrating wastewater treatment and cultural ecological services in constructed wetland systems (Ghermandi and Fichtman, 2015). However, some point out that the extent (and form) of participation that evolves is (inversely) related to the degree of centralised control imposed at national level (Benson *et al*., 2013a,b). This may be a particular issue for European countries that are implementing EU directives.

The value of the participatory approach, now an important element of most EU actions, has also been appreciated outside Europe. For instance, the preference for preventing water pollution at source, rather than during end-of-pipe treatment, has been appreciated in the USA for some time. For example, in New York City, stakeholder involvement and participation were found to be the most cost-effective ways of addressing the deterioration in the quality of the water supply from the

Catskills Mountains, due to the economically induced intensification of agriculture in that catchment (Brils *et al*., 2015). A decision was made to invest in “natural” capital on the basis of the expectation that a return of a greater value (i.e. a profit) would be achieved. An investment of less than US $1.5 billion was spent on addressing the Catskills water supply, and it was expected that this would produce a return of over US $6 billion, which is equivalent to the savings in capital costs of a plant to treat the polluted water (Chichilnisky and Heal, 1998). Much of the expenditure was used to buy land in the catchment and restrict the activities taking place on that land and thus the potential for pollution.

7.1 Stakeholders and Perceptions

The scale at which humans as organisms perceive landscapes, what we term the perceptible realm, is particularly important because this is the scale at which humans intentionally change landscapes, and these changes affect environmental processes (Gobster *et al*., 2007). Åberg and Tapsell (2013) showed that people’s aesthetic preferences are strongly related to the ecological quality of the rehabilitated river reach. However, people have a preference for what appears natural to them, which might not always be in accordance with expert opinions on what provides ecological quality (Nassauer *et al*., 2001; Gobster *et al*., 2007).

7.2 Stakeholders and Trade-offs

As outlined in section 2.2, trade-offs can occur when the provision of one ecosystem service is reduced as a consequence of increased use of another. However, trade-offs also occur when one ecosystem service is preferred by stakeholders at the expense of other ecosystem services (Rodriguez *et al*., 2006). Trade-offs can also occur among stakeholders with competing interests (Wells and McShane 2004; McShane *et al*., 2011; Howe *et al*., 2014). This is a direct result of different groups of stakeholders deriving well-being from different ecosystem services and, consequently, valuing different management options for maximum delivery of the ecosystem service(s) of interest (Howe *et al*., 2014). These trade-offs between stakeholders can be continuous and ubiquitous, but also vary spatially and temporally. For example, there may be competing interests between local and national stakeholders, or between stakeholders at different points along a catchment, or between varying seasonal interests among different stakeholders when a particular ecosystem service is only available, or is more valued, at a specific time of year.

8 The Role of Ecosystem Services within the Legislative and Policy Framework

Ecosystem services are often under-represented in policy decisions despite their importance for society's well-being, owing mainly to market failures and lack of awareness (Locatelli *et al*., 2011). Outlined here are a number of the legislation/policy targets that do require consideration of ecosystem services and the ecological components underpinning them. The first initiative, introduced in March 2010 by the European Council, committed to a new vision for biodiversity and a target of "Halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss" (EC, 2010). A second target for 2050, states that: "By 2050, European Union biodiversity and the ecosystem services it provides – its natural capital – are protected, valued and appropriately restored for biodiversity's intrinsic value and for their essential contribution to human wellbeing and economic prosperity, and so that catastrophic changes caused by the loss of biodiversity are avoided" (EC, 2010). However, in order to meet these targets, an improved and scientifically robust knowledge of ecosystems and the services they provide is required. Therefore, ecosystem services have been included as a core concept in Action 5 of the EU Biodiversity Strategy to 2020. This requires "Member States to map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020" (EC, 2013c). While the EU Biodiversity Strategy does not specifically define or categorise ecosystem services, it will incorporate the findings of the recent version of CICES (version 4.3) undertaken by the EEA (Haines-Young and Potschin, 2013).

Ireland's National Biodiversity Action Plan (NPWS, 2011) lists the strategies in place to safeguard, monitor and assess biodiversity, and includes a range of responsibilities relevant to water and the protection of aquatic species. These also include obligations under the WFD, which is a key piece of EU legislation which states that Member States must improve and sustainably maintain water quality; the emphasis of this directive is on maintenance/restoration of good ecological quality. As outlined by Hering *et al*. (2015), there are four elements to water resources management in Europe: river basin district (RBD) planning, status assessment, risk assessment, and economic analysis of the costs and benefits of management. The final element mentioned relates in particular to the ecosystem services framework and can be considered under the ecosystem service cascade framework (e.g. see Figure 1.2), which highlights the role of governance and policy (Haines-Young and Potschin, 2010a).

A number of other policy documents emphasise the need to address water quality and quantity issues, and consider ecosystem service protection and valuation. For example, *Food Harvest 2020* (DAFM, 2010) highlights the importance of water quality protection and improvement for sustainable food production. Because an integrated approach to all legislative requirements is sought, the Floods Directive (FD) and nitrate regulations are also of relevance. The European Commission communication "A Blueprint to Safeguard Europe's Water" (EC, 2012) strives to halt the deterioration of water quality, the associated loss of biodiversity and the degradation of ecosystem services in the EU by 2020. This blueprint also refers to the need to further implement the concept of payment for ecosystem services (PES). It recognises the complexity of the interactions between freshwater ecosystem stressors and that stakeholder involvement is needed to achieve improvement. Other recent developments include the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). This is an international initiative, which commenced in April 2012, that aims to be an independent platform for members of the United Nations (UN) to provide information on the assessment of ecosystems and ecosystem services for both the policy and research communities. The IPBES work programme combines four objectives:

1. to develop the conceptual framework to combine knowledge generation, assessment, policy support tools and capacity building;

2. to provide methodological consistency of deliverables addressing the production and integration of assessments, including on a regional/sub-regional scale and globally;
3. to provide a thematic assessment of pollination, land degradation, invasive species and sustainable land use, along with policy support tools for scenario analysis and modelling and for the diverse conceptualisation of values of biodiversity;
4. to deliver a catalogue of recent assessments and information, data management and policy support tools, methodologies, and a review of guidance and procedures for future development.

8.1 Incorporating the Ecosystem Services Framework into Water Resources Protection and Management as Required by the WFD

The roots of European actions related to ecosystem services can be traced back to the UN MEA (2005), initiated in 2001. One of the knowledge gaps identified in that assessment was that "little information is available about the economic value of non-marketed services" and that this is important because "the costs of the depletion of these services are rarely tracked in national economic accounts". The revised European approach to sustainable development is guided by the MEA (2005) and is founded on four pillars: environmental protection, social equity and cohesion, economic prosperity and meeting the EU's international obligations and responsibilities (EC, 2006). This document sets out the key challenges and one of these is "conservation and management of natural resources" with the overall objective "to improve management and avoid overexploitation of natural resources, recognising the value of ecosystem services". The word "recognised" is particularly weak in this context and the text falls short of stipulating that ecosystems services must be directly taken into account when decisions are made. While preservation of biodiversity and improving IWRM are identified as actions that support the objective (Bruen, 2009), the role of ecosystem services valuation is not explicitly identified. However, the EU's own guidelines for impact assessments, which must precede each major action, specify that the environmental, social and economic aspects of any potential impact must be assessed (EC, 2009). Ecosystem services valuations must be part of this, and Diehl *et al*. (2015) argued that such valuations would help to, inter alia, (1) improve the set of indicators used in decision making, as well as (2) facilitate the upscaling and downscaling of benefits and costs needed to compare a diverse range of potential actions, and (3) assist with the integration of the perception of stakeholders within a decision process.

The European Commission's communication on green infrastructure (EC, 2013b) emphasises the important role of natural capital and the value of the ecosystem services concept for providing an integrated and balanced perspective. Some of the challenges identified are (1) scale issues and (2) the need for consistent and reliable data on which to base valuations. The specific benefits of incorporating the ecosystem services framework into the implementation of both the WFD and the FD were identified by COWI A/S (2014) and Blackstock *et al*. (2015) and further developed as follows:

- it assists with the delivery of the objectives of the WFD with the wider policy imperatives of sustainability, integration and subsidiarity, helping it to achieve its original ambitions;
- it helps to illustrate how human well-being is dependent on ecological health, widening the focus from good ecological status as an end in itself to showing how it supports societal goals;
- it allows the proper assessment and communication of the benefits and co-benefits of implementing the WFD and FD, explaining the trade-offs involved in selecting cost-effective measures;
- it enables a more comprehensive evaluation of the benefits and costs of measures to improve water quality;
- it avoids unintended impacts of measures on other benefits (not directly associated with the measure); this is facilitated by the broad overview provided by the ecosystem services framework;
- it ensures a better understanding of who gains and loses from specific measures, as the ecosystem services framework requires the identification of stakeholders, in addition to widening and deepening the engagement of stakeholders;
- it assists with planning the integrated implementation of multiple directives, as it can describe the impact of each in terms of a single set of descriptors (the ecosystems services); similarly, these descriptors can provide a unifying basis for various strategies (e.g. the EU Biodiversity Strategy).

The steps involved are (1) the identification of the benefits/costs; (2) the quantification of the benefits/costs; and (3) the valuation of the benefits/costs. The ecosystem services framework can be seen to provide a more quantitative basis for managing and assessing the benefits of implementing the WFD and FD, and fits into the theoretical chain of knowledge outlined by Mooney (2009) in Chapter 4 regarding the incorporation of ecosystem services into policy. This is reinforced by an in-depth study by the European Directorate for Research and Innovation and ONEMA on the issues arising from the implementation of the WFD (Reyjol *et al*., 2014). This science–policy interface activity produced a large number of recommendations, including the need to address the current disconnection between "good ecological status" and specific ecosystem services, a gap also identified by others (Vlachopoulou *et al*., 2014). This requires a better understanding of ecosystem processes and an increase in awareness among policymakers and managers of the concept of ecosystem services. The issues of spatial scale and fit are important (Moss, 2012), as is the need for co-operation between organisations that operate at different scales, each of which has some responsibility for implementing the WFD (Huesker and Moss, 2015) or water management (Lavenus and Chazot, 2015), particularly in the context of how the catchment as the unit of management fits in with established local regional institutions and political jurisdictions. This was echoed by Bastian *et al*. (2012) who identified the critical spatial, temporal and dimensional aspects of ecosystem services and linked these with specific aspects of the WFD.

In specific application areas, attempts have been made to develop a unified and agreed set of ecosystem services indicators, for example for biodiversity (Maes *et al*., 2016). Nevertheless, there is a realisation that a single ecosystem services framework may not be applicable in all cases and that its methodology must be tailored to suit circumstances in individual EU countries (COWI A/S, 2014). Issues of scale also arise in the context of individual indicators, for example Kelly (2013) pointed out that current European phytobenthos metrics may be appropriate for the site-specific chemical status of water bodies, suited to the WFD, but may not encompass the risk to trophic levels or ecosystem services. A balance is required between managed and free-flowing rivers, both of which are assessed in the framework of ecosystems services (Auerbach *et al*., 2014). The greater focus on services (positive) as opposed to disservices (negative) has been criticised by Sandbrook and Burgess (2015), who pointed out that both services and disservices to humanity may be provided by the same species (see section 2.2 for more details). The need for a multi-criteria approach (or at least an extended cost–benefit approach) is acknowledged, as such an approach allows the incorporation of a mixture of quantitative and qualitative criteria into policy decisions.

The case of inland wetlands in Spain, of which 27 different types were identified by the Spanish strategic study for the conservation and rational use of wetlands (Ministerio de Medio Ambiente, 2000), highlights the difficulties caused by a lack of specific information with regard to applying the ecosystem services framework with WFD implementation (de la Hera *et al*., 2011). The authors of this study highlighted the difficulties encountered when the character of wetlands change; for example, in Tablas de Daimiel National Park, what was once a groundwater-dependent wetland changed to become a source of aquifer recharge when local groundwater levels dropped because of increased extractions for irrigated agriculture. Difficulties arose with regard to the identification of all of the services and goods provided by the multiplicity of wetland types. Similarly, Blackstock *et al*. (2015) highlighted several general challenges surrounding the ecosystem services framework and its inclusion in the WFD. They include:

- supporting trans-disciplinary decision making about a social–ecological system rather than technical decisions about individual environmental issues;
- placing values (monetary and non-monetary) on the full range of ecosystem services, which is difficult and expensive;
- taking account of different timescales for human and ecological processes, and reconciling the different geographical scales for assessing ecology and reporting on progress;
- providing additional monitoring, methodologies and mapping, which, again, may be expensive.

The Irish EPA's Catchments Unit is driven mainly by the WFD (i.e. an ecological focus) and promotes *integrated water management*, for which the catchment is the major unit of analysis (Daly, 2015a,b). Since services provided by the environment have both biotic and abiotic components, Daly (2015a) recommends combining ecosystem services and geosystem services

with human/social system services into a single entity, called “catchment services” (see section 1.1.2). This would have a significant practical impact on promoting consistency in valuation and decision methods across the field of ecosystem, geosystem and human/social system services. Nevertheless, the use of the ecosystem service-based approach on its own in catchment management and planning is well suited to linking biodiversity, ecosystem function and human well-being (Blackstock *et al*., 2015).

8.2 An Alternative Policy Framework

One proposed approach for incorporating ecosystem services into a policy framework has been suggested by Kelble *et al*. (2013). For this approach, the authors suggested merging the DPSIR conceptual model (see section 4.1 and Figure 4.3) with ecosystem services giving rise to an “Ecosystem Based Management – Driver, Pressure, State, Ecosystem Service, Response” (EBM-DPSER) conceptual model (Figure 8.1).

The DPSIR model is used to analyse and assess environmental issues, and brings together different scientific disciplines, environment managers and stakeholders in order to develop solutions for sustainable development; ultimately, it is designed to inform ecosystem management in order to protect ecosystems from human impacts (Gari *et al*., 2015). In contrast, the EBM-DPSER model is designed to inform ecosystem management for the benefit of humans (Figure 8.2).

Kelble *et al*. (2013) stated that linking the pressures (P) to states (S) and then to ecosystem services (E) allows the qualitative assessment of “the cumulative impacts of pressures upon ecosystem services” and “captures the direct and indirect effect of multiple human uses on ecosystem services”. This also highlights the loss of ecosystem services to human society (Kelble *et al*., 2013). The responses (R) within the EBM-DPSER model incorporate the human actions (management, policy or other) motivated by changes in the condition in the environment (state) or in the ecosystem services provided (Kelble *et al*., 2013). Responses can affect drivers, pressures, states or ecosystem services and represent a mechanism for feedback by society. Ultimately, this approach emphasises the extent to which society relies upon and benefits from ecosystems by highlighting the role of ecosystem services (Kelble *et al*., 2013). In addition, the approach of the EBM-DPSER model could help to enhance and bridge the communication gap between social scientists, biophysical scientists, resource managers and policymakers, thus aiding ecosystem and catchment-based management.

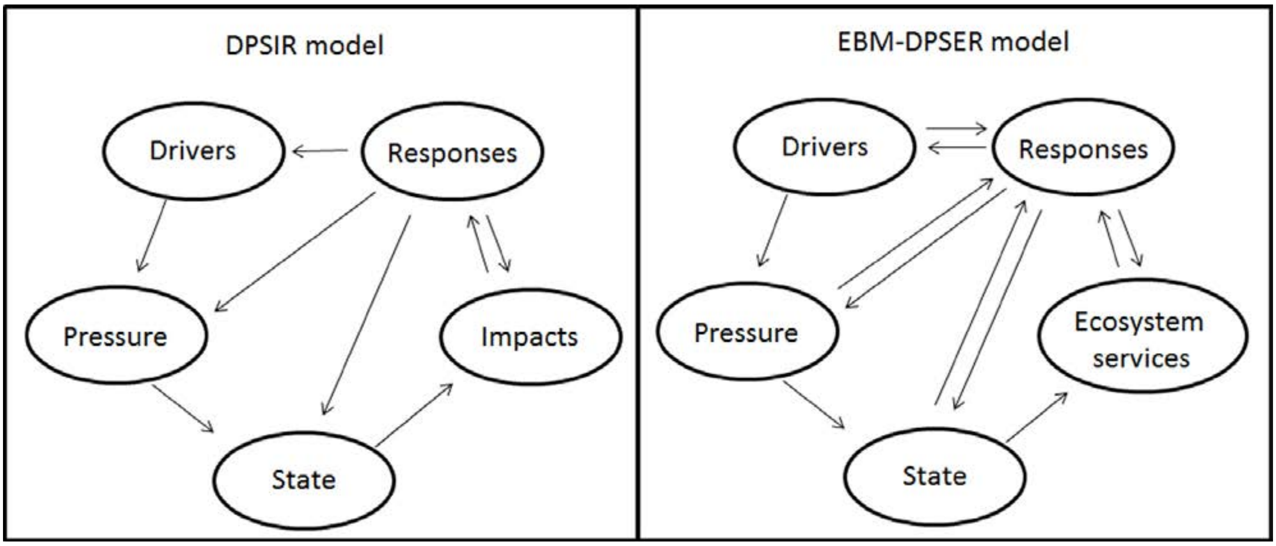

Figure 8.1. A comparison between the DPSIR and EBM-DPSER conceptual models. Figure adapted from Kelble *et al*. (2013).

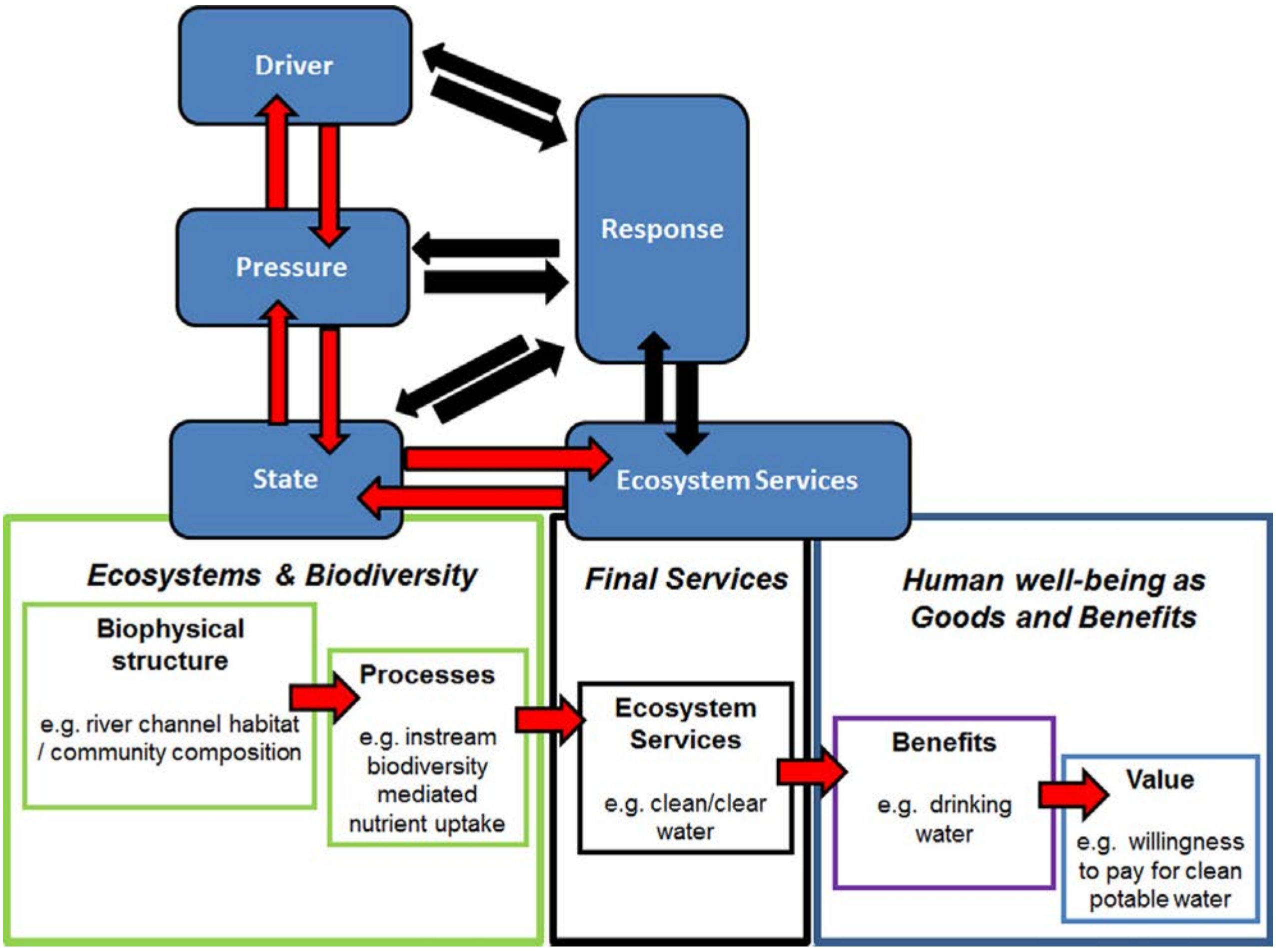

Figure 8.2. The EBM-DPSER conceptual model superimposed on the ecosystem cascade model outlined in Figure 1.2. The DPSIR model (see Figure 4.3) was modified by replacing the impacts module with ecosystem services, thereby facilitating a more complete representation of ecosystem interactions including those with human society and the associated feedbacks.

9 Conclusions

Regardless of the terminology and approach used, “the process of identifying nature’s values is not to be taken as an end in itself. It should be treated as a means to better communicate and take account of nature’s importance in policy- and decision making, with particular regard to human well-being and to the conservation of natural commons for reasons of inter- and intra-generational equity” (Sukhdev *et al*., 2014).

References

Åberg, E.U. and Tapsell, S., 2013. Revisiting the River Skerne: the long-term social benefits of river rehabilitation. *Landscape and Urban Planning* 11: 94–103.

Adamowicz, W., Louviere, J. and Williams, M., 1994. Combining revealed and stated preference methods for valuing environmental amenities. *Journal of Environmental Economics and Management* 26: 271–292.

Alexander, R.B., Boyer, E.W., Smith, R.A. *et al*., 2007. The role of headwater streams in downstream water quality. *Journal of the American Water Resources Association* 43: 41–59.

Amigues, J.P. and Chevassus-au-Louis, B., 2011. *Assessing the Ecological Services of Aquatic Environments: Scientific, Political and Operational Issues*. ONEMA. Available online: http://www.onema.fr/IMG/EV/cat7a-ecological-services.html (accessed 10 December 2015).

Anderson, T., Carstensen, J. and Duarte, C.M., 2008. Ecological thresholds and regime shifts: approaches to identification. *Trends in Ecology and Evolution* 24: 49–57.

Andreopoulos, D., Damigos, D., Comiti, F. *et al*., 2015. Estimating the non-market benefits of climate change adaptation of river ecosystem services: a choice experiment application in the Aoos basin, Greece. *Environmental Science and Policy* 45: 92–103.

Auerbach, D.A., Deisenroth, D.B., McShane, R.R. *et al*., 2014. Beyond the concrete: accounting for ecosystem services from free-flowing rivers. *Ecosystem Services* 10: 1–5.

Balmford, A., Fisher, B., Green, R.E. *et al*., 2011. Bringing ecosystem services into the real world: an operational framework for assessing the economic consequences of losing wild nature. *Environmental and Resource Economics* 48: 161–175.

Balvanera, P., Siddique, I., Dee, L. *et al*., 2014. Ecosystem services: current uncertainties and the necessary next steps. *BioSciences* 64: 49–57.

Barbier, E.B., 2007. Valuing ecosystem services as productive inputs. *Economic Policy* 22: 177–229.

Bassar, R.D., Marshall, M.C., López-Sepulcre, A. *et al*., 2010. Local adaptation in Trinidadian guppies alters ecosystem processes. *Proceedings of the National Academy of Sciences of the United States of America* 107, 3616–3621.

Bastian, O., Grunewald, K. and Syrbe, R.-U., 2012. Space and time aspects of ecosystem services, using the example of the EU Water Framework Directive. *International Journal of Biodiversity Science, Ecosystems Services and Management* 8: 5–16.

Bateman, I., Carson, R., Day, B. *et al*., 2003. *Guidelines for the Use of Stated Preference Techniques for the Valuation of Preferences for Non-market Goods*. Edward Elgar, Cheltenham.

Bateman, I.J., Day, B.H., Georgiou, S. *et al*., 2006. The aggregation of environmental benefit values: welfare measures, distance decay and total WTP. *Ecological Economics* 60: 450–460.

Bateman, I.J., Brouwer, R., Ferrini, S. *et al*., 2011. Making benefit transfers work: deriving and testing principles for value transfers for similar and dissimilar sites using a case study of the non-market benefits of water quality improvements across Europe. *Environmental Resource Economics* 50: 365–387.

Bennett, E.M., Peterson, G.D. and Gordon, L.J., 2009. Understanding relationships among multiple ecosystem services. *Ecology Letters* 12: 1–11.

Benson, D., Jordan, A., Cook, H. *et al*., 2013a. Collaborative environmental governance: are watershed partnerships swimming or are they sinking? *Land Use Policy* 30: 748–757.

Benson, D., Jordan, A. and Smith, L., 2013b. Is environmental management really more collaborative? A comparative analysis of putative “paradigm shifts” in Europe, Australia, and the United States. *Environment and Planning A* 45: 1695–1712.

Berbés-Blázquez, M., Oestreicher, J.S., Mertens, F. *et al*., 2014. Ecohealth and resilience thinking: a dialog from experiences in research and practice. *Ecology and Society* 19: 24.

Biggs, R., Schlüter, M., Biggs, D. *et al*., 2012. Toward principles for enhancing the resilience of ecosystem services. *Annual Review of Environment and Resources* 37: 421–448.

Birol, E., Koundouri, P. and Kountouris, Y., 2008. *Using the Choice Experiment Method to Inform Flood Risk Reduction Policies in the Upper Silesia region of Poland.* MPRA (Munich Personal RePEc Archive) Paper No. 38426. Available online: http://wpa.deos.aueb.gr/docs/2009.CambridgeUniversityPress.pdf (accessed 2 December 2015).

Blackstock, K.L, Martin-Ortega, J. and Spray, C.J., 2015. Implementation of the European Water Framework Directive. In Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. and Khan, S. (eds), *Water Ecosystem Services*. Cambridge University Press, Cambridge, UK.

Blindow, I., Anderson, G., Hargeby, A. *et al*., 1993. Long-term pattern of alternative stable states in two shallow eutrophic lakes. *Freshwater Biology* 30: 159–167.

Boyd, J. and Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63: 616–626.

Braat, L.C. and de Groot, R., 2012. The ecosystem services agenda: bridging the worlds of natural science and economics, conservation and development, and public and private policy. *Ecosystem Services* 1: 4–15.

Brander, L.M., Florax, J.G.H. and Vermaat, J.E., 2006. The empirics of wetland valuation: a comprehensive summary and meta-analysis of the literature. *Environmental and Resource Economics* 33: 223–250.

Brauman, K., van der Meulen, S. and Brils, J., 2014. Ecosystem services in river basin management. Risk-informed management of European river basins. In Brils, J., Brack, W., Muller, D. *et al*. (eds), *Risk-Informed Management of European River Basins*. Springer, Berlin.

Brils, J., Appleton, A., van Everdingen, N. *et al*., 2015. Key factors for successful application of ecosystem services-based approaches to water resources management: the role of stakeholder participation. In Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. and Khan, S. (eds), *Water Ecosystem Services*. Cambridge University Press, Cambridge, UK.

Briske, D.D., Washington-Allen, R.A., Johnson, C.R. *et al*., 2010. Catastrophic thresholds: a synthesis of concepts, perspectives and applications. *Ecology and Society* 15: 37.

Brouwer, R. and Panagiotis, B., 2007. *Economic Assessment of Environment and Resource Costs in the Water Framework Directive (AquaMoney*). Available online: http://ecologic.eu/1776 (accessed 16 June 2016).

Brown, T.C., Bergstrom, J.C. and Loomis, J.B., 2007. Defining, valuing and providing ecosystem goods and services. *Natural Resources Journal* 47: 329–376.

Bruen, M., 2009. Hydrology and the Water Framework Directive in Ireland. *Biology and Environment: Proceedings of the Royal Irish Academy* 109B: 207–220.

Bruen, M., Kelly, M., Magette, W., *et al*., 2010. *WINCOMS: Water Framework Directive – Integration, Negotiation and Communication of Optimal Measures with Stakeholders*. WINCOMS project final report. EPA, Johnstown Castle, Ireland.

Bryan, B.A., Raymond, C.M., Crossman, N.D. *et al*., 2010. Targeting the management of ecosystem services based on social values: where, what and how? *Landscape and Urban Planning* 97: 111–122.

Bullock, A. and Acreman, M.C., 2003. The role of wetlands in the hydrological cycle. *Hydrology and Earth System Sciences* 7: 75–86.

Bullock, J.M., Aronson, J., Newtown, A.C. *et al*., 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. *Ecology and Evolution* 26: 541–549.

Burkhard, B., Kandziora, M., Hou, Y. *et al*., 2014. Ecosystem service potentials, flows and demands – concepts for spatial localisation, indication and quantification. *Landscape Online* 34: 1–32.

Burkhard, B., Kroll, F., Nedkov, S. *et al*., 2012. Mapping ecosystem service supply, demand and budgets. *Ecological Indicators* 21: 17–29.

Burton, M., Marsh, S. and Patterson, J., 2007. Community attitudes towards water management in the Moore Catchment, Western Australia. *Agricultural Systems* 92: 157–178.

Callanan, M.C., Baars, J-R. and Kelly-Quinn, M., 2008. Critical influence of seasonal sampling on the ecological quality assessment of small headwater streams. *Hydrobiologia* 610: 245–255.

Cardinale, B.J., Duffy, J.E., Gonzalez, A. *et al*., 2012. Biodiversity loss and its impact on humanity. *Nature* 486: 59–68.

Carpenter, S.R., 2003. Regime shifts in lake ecosystems: pattern and variation. In Kinne, O. (ed.), *Excellence in Ecology*. International Ecology Institute, Oldendorf, Germany.

Castro, A.J., Martin-Lopez, B., Lopez, E. *et al*., 2015. Do protected areas networks ensure the supply of ecosystem services? Spatial patterns of two nature reserve systems in semi-arid Spain. *Applied Geography* 60: 1–9.

CBD (Convention on Biological Diversity), 2000. Ecosystem approach: further conceptual elaboration. In *Recommendation V/10 adopted by the Subsidiary Body on Scientific, Technical and Technological Advice at its Fifth Meeting*. Fifth meeting of the Subsidiary Body on Scientific, Technical and Technological Advice, 31 January–4 February 2000, Montreal, Canada. Available online: https://www.cbd.int/doc/recommendations/sbstta-05/full/sbstta-05-rec-en.pdf (accessed 14 June 2016).

CBD (Convention on Biological Diversity), 2004. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Seventh Meeting – VII/11: Ecosystem Approach*. Conference of the Parties to the Convention on Biological Diversity, seventh meeting, 9–20 and 27 February 2004, Kuala Lumpur, Malaysia. Available online: http://www.cbd.int/doc/decisions/cop-07/cop-07-dec-11-en.doc (accessed 3 May 2016).

CBD (Convention on Biological Diversity), 2006. *Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity at its Eighth Meeting –VIII/9: Implications of the Findings of the Millennium Ecosystem Assessment*. Conference of the Parties to the Convention on Biological Diversity, eighth meeting, 20–31 March 2006, Curitiba, Brazil. Available online: http://www.cbd.int/doc/decisions/COP-08/cop-08-dec-09-en.doc (accessed 14 June 2016).

CDM (Camp Dresser & McKee), 2004. *Economic Analysis of Water Use in Ireland: Final Report*. Submitted to the Department of the Environment, Heritage and Local Government, 212 pp. Available online: http://www.wfdireland.ie/docs/35_Economics/Economic%20Analysis%20of%20Water%20use.pdf (accessed 2 December 2015).

Chan, K.M.A., Shaw, M.R., Cameron, D.R. *et al*., 2006. Conservation planning for ecosystem services. *PLoS Biology* 4: e379.

Chapin, F.S., Zavaleta, E.S., Eviner, V.T. *et al*., 2000. Consequences of changing biodiversity. *Nature* 405: 234–242.

Chichilnisky, G. and Heal, G., 1998. Economic returns from the biosphere. *Nature* 391: 629–630.

Christie, M. and Azevedo, C., 2009. Testing the consistency between standard contingent valuation, repeated contingent valuation, and choice experiments. *Journal of Agricultural Economics* 60: 154–170.

Christie, M. and Rayment, M., 2012. An economic assessment of the ecosystem service benefits derived from the SSSI biodiversity conservation policy in England and Wales. *Ecosystem Services* 1: 70–84.

Christie, M., Hanley, N., Warren, J.M. *et al*., 2006. Valuing the diversity of biodiversity. *Ecological Economics* 58: 304–317.

Christie, M., Fazey, I., Cooper, R. *et al*., 2008. *An Evaluation of Economic and Non-economic Techniques for Assessing the Importance of Biodiversity to People in Developing Countries*. DEFRA, London, UK.

Christie, M., Fazey, I., Cooper, R. *et al*., 2012. An evaluation of monetary and non-monetary techniques for assessing the importance of biodiversity and ecosystem services to people in countries with developing economies. *Ecological Economics* 83: 69–80.

CICES (Common International Classification of Ecosystem Services), 2013. CICES-V4-3. Available online: http://cices.eu/ (accessed 2 December 2015).

Cornell, S., Berkhout, F., Tuinstra, W. *et al*., 2013. Opening up knowledge systems for better responses to global environmental change. *Environmental Science and Policy* 28: 60–70.

Costanza, R., 1991. *Ecological Economics: The Science and Management of Sustainability*. Columbia University Press, New York, NY.

Costanza, R., 2008. Ecosystem services: multiple classification systems are needed. *Biological Conservation* 141: 350–352.

Costanza, R., d'Arge, R., de Groot, R. *et al*., 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.

COWI A/S, 2014. *Support Policy Development for Integration of an Ecosystem Services Approach with WFD and FD Implementation. Towards Practical Guidelines to Support River Basin Planners*. COWI A/S, Kongens Lyngby, Denmark. Available online: https://circabc.europa.eu/sd/a/95c93149-0093-473c-bc27-1a69cface404/Ecosystem%20service_WFD_FD_Main%20Report_Final.pdf (accessed 2 December 2015).

Cowling, R.M., Egoh, B., Knight, A.T. *et al*., 2008. An operational model for mainstreaming ecosystem services into implementation. *Proceedings of the National Academy of Sciences of the United States of America* 105: 9483–9488.

Cummins, K.W., 1992. Catchment characteristics and river ecosystems. In Boon, P.J., Calow, P. and Petts, G.E. (eds), *River Conservation and Management*. Wiley, Chichester, United Kingdom, 125–135.

Curtis, J.A., 2002. Estimating the demand for salmon angling in Ireland. *The Economic and Social Review* 33: 319–332.

Curtis, J.A., 2003. Demand for water-based leisure activity. *Journal of Environmental Planning and Management* 46: 65–77

DAFM (Department of Agriculture, Food and the Marine), 2011. *Food Harvest 2020. A vision for the Irish agri-food and fisheries.* Available online: https://www.agriculture.gov.ie/media/migration/foodindustrydevelopmenttrademarkets/foodharvest2020/2020FoodHarvestEng240810.pdf (accessed 1 June 2016).

Daily, G.C., 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press, Washington, DC.

Daly, D., 2015a. *The Concept of Catchment Services in the Context of Water Management and WFD Implementation*. Discussion paper. EPA, Dublin, Ireland.

Daly, D., 2015b. The catchment services concept – A means of connecting and progressing Water Framework Directive and biodiversity requirements in the context of sustainable intensification of agriculture. In Ó hUallacháin, D. and Finn, J.A. (eds), *Farmland Conservation with 2020 Vision*. Teagasc, Wexford, Ireland.

DEFRA (Department for Environment, Food & Rural Affairs), 2007. *An Introductory Guide to Valuing Ecosystem Services.* DEFRA, London, UK. Available online: https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/69192/pb12852-eco-valuing-071205.pdf (accessed 2 December 2015).

de Groot, R.S., Wilson, M.A. and Boumans, R.M.J., 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics* 41: 393–408.

de la Hera, A., Fornés, J.M. and Bernués, M., 2011. Ecosystem services of inland wetlands from the perspective of the EU Water Framework Directive implementation in Spain. *Hydrological Sciences Journal* 56: 1656–1666.

Díaz, S., Tilman, D. and Fargione, J., 2005. Biodiversity regulation of ecosystem services. In Hassan, R., Scholes, R., and Ash, N. (eds), *Ecosystems and Human Well-Being. Current State and Trends – Findings of the Condition and Trends Working Group of the Millennium Ecosystem Assessment*. Island Press, Washington, DC.

Diehl, K., Burkhard B. and Jacob K., 2015. Should the ecosystems services concept be used in European Commission impact assessment? *Ecological Indicators* 61: 6–17.

DKM, ESRI (Economic and Social Research Institute) and Aquavarra, 2004. *Economic Evaluation of Water Supply and Waste Water Projects – Cost–Benefit Analysis Methodology Paper*. Report for the Department of the Environment, Heritage and Local Government. Available online: http://studylib.net/doc/5897321/economic-evaluation-of-water-supply-and-waste-water-projects (accessed 2 December 2015).

Doherty, E., Murphy, G., Hynes, S. *et al*., 2014. Valuing ecosystem services across water bodies: results from a discrete choice experiment. *Ecosystem Services* 7: 89–97.

Durance, I., Bruford, M.W., Chalmers, R. *et al*., 2016. The challenges of linking ecosystem services to biodiversity: lessons from a large-scale freshwater study. *Advances in Ecological Research* 54: 87–134.

Eastwood, A., Brooker, R., Irvine, R.J. *et al*., 2016. Does nature conservation enhance ecosystem services delivery? *Ecosystem Services* 17: 152–162.

EC (European Commission), 2006. *Review of the EU Sustainable Development Strategy (EU SDS) – Renewed Strategy*. Available online: https://www.etuc.org/sites/www.etuc.org/files/st10117.en06.pdf (accessed 2 December 2015).

EC (European Commission), 2009. *Impact Assessment Guidelines*. Available online: http://ec.europa.eu/smart-regulation/impact/commission_guidelines/docs/iag_2009_en.pdf (accessed 2 December 2015).

EC (European Commission), 2010. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions "Options for an EU vision and target for biodiversity beyond 2010". Available online: http://ec.europa.eu/environment/nature/biodiversity/policy/pdf/communication_2010_0004.pdf (accessed 1 July 2016).

EC (European Commission), 2011. *EU Biodiversity Strategy to 2020*. Available online: http://ec.europa.eu/environment/pubs/pdf/factsheets/biodiversity_2020/2020%20Biodiversity%20Factsheet_EN.pdf (accessed 2 December 2015).

EC (European Commission), 2012. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions "A Blueprint to Safeguard Europe's Water Resources". Available online: http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52012DC0673&from=EN (accessed 2 December 2015).

EC (European Commission), 2013a. *System of Environmental-Economic Accounting 2012: Experimental Ecosystem Accounting*. Available online: http://unstats.un.org/unsd/envaccounting/eea_white_cover.pdf (accessed 2 December 2015).

EC (European Commission), 2013b. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions "Green Infrastructure (GI) – Enhancing Europe's Natural Capital". Available online: http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:52013SC0155&from=EN (accessed 2 December 2015).

EC (European Commission), 2013c. *Mapping and Assessment of Ecosystems and their Services: An Analytical Framework for Ecosystem Assessments under Action 5 of the EU Biodiversity Strategy to 2020*. Available online: http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/MAESWorkingPaper2013.pdf (accessed 1 July 2016).

EC (European Commission), 2014. *Mapping and Assessment of Ecosystems and their Services: Indicators for Ecosystem Assessments under Action 5 of the EU Biodiversity Strategy to 2020*. Technical Report – 2014 – 080. Available online: http://ec.europa.eu/environment/nature/knowledge/ecosystem_assessment/pdf/2ndMAESWorkingPaper.pdf (accessed 2 December 2015).

EEA (European Environment Agency), 2015. *European Ecosystem Assessment — Concept, Data and Implementation. Contribution to Target 2 Action 5 Mapping and Assessment of Ecosystems and their Services (MAES) of the EU Biodiversity Strategy to 2020*. EEA Technical Report No 6/2015. Available online: http://www.eea.europa.eu/publications/european-ecosystem-assessment (accessed 2 December 2015).

Eftec, 2006. *Valuing our Natural Environment: Final Report*. DEFRA, London, UK. Available online: http://earthmind.net/rivers/docs/ukdefra-eftec-valuing-our-natural-environment.pdf (accessed 2 December 2015).

Egoh, B.N., Knight, A., Cowling, R.M. *et al*., 2007. Integrating ecosystem services into conservation assessments: a review. *Ecological Economics* 63: 714–721.

Egoh, B., Reyers, B., Rouget, M. *et al*., 2008. Mapping ecosystem services for planning and management. *Agriculture, Ecosystems and Environment* 127: 135–140.

Ehrlich, P.R. and Ehrlich. A.H., 1981. *Extinction: The Causes and Consequences of the Disappearance of Species.* Random House, New York, NY.

Ehrlich, P. and Mooney. H., 1983. Extinction, substitution, and ecosystem services. *BioScience* 33: 248–254 Eigenbrod, F., Anderson, B.J., Armsworth, P.R. *et al*., 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. *Proceedings of the Royal Society B – Biological Sciences* 276: 2903–2911.

Eigenbrod, F., Anderson, B.J., Armsworth, P.R. *et al*., 2009. Ecosystem service benefits of contrasting conservation strategies in a human-dominated region. *Proceedings of the Royal Society B – Biological Sciences* 276: 2903–2911.

Elmqvist, T., Folke, C., Nyström, M. *et al*., 2003. Response diversity, ecosystem change, and resilience. *Frontiers in Ecology and the Environment* 1: 488–494.

EPA (Environmental Protection Agency), 2010a. *Contaminant Movement and Attenuation along Pathways from the Land Surface to Aquatic Receptors – A Review.* EPA STRIVE Report Series No. 56. Available online: http://www.epa.ie/pubs/reports/research/water/STRIVE_56_Archbold_ContaminantsPathwaysReview_web.pdf (accessed 2 December 2015).

EPA (Environmental Protection Agency), 2010b. *Water Quality in Ireland 2007–2009*. EPA, Johnstown Castle, Ireland.

EPA (Environmental Protection Agency), 2012a. *Benefit Transfer for Irish Water.* STRIVE Report Series No. 94. EPA, Johnstown Castle, Ireland.

EPA (Environmental Protection Agency), 2012b. *Ireland's Environment: An Assessment*. EPA, Johnstown Castle, Ireland.

EPA (Environmental Protection Agency), 2014. *Are We Willing to Pay for Good River Water Quality? Willingness to Pay for Achieving Good Status Across Rivers in the Republic of Ireland*. EPA, Johnstown Castle, Ireland. Available online: https://www.epa.ie/pubs/reports/research/water/EPA_Research_Report_%20129.pdf (accessed 14 April 2016).

EPA (Environmental Protection Agency), 2015. *Water Quality in Ireland 2010–2012*. EPA, Johnstown Castle, Ireland.

Fahy, E., 1995. *The Brown Trout in Ireland*. *A Fragile History*. Immel Publishing, London, UK.

Feeley, H.B., Bruen, M., Blacklocke, S. *et al*., 2013. A regional examination of episodic acidification response to reduced acidic deposition and the influence of plantation forests in Irish headwater streams. *Science of the Total Environment* 443: 173–183.

Feeley, H.B., Bruen, M., Bullock, C., *et al*. (2016). Synthesis of the current knowledge on Irish freshwater resources in the context of ecosystem services. Report to EPA (Environmental Protection Agency), Johnstown Castle, Ireland.

Field, C., Behrenfeld, M., Randerson, J. *et al*., 1998. Primary production of the biosphere: integrating terrestrial and oceanic components. *Science* 281: 237–240.

Fish, R.D., 2011. Environmental decision making and an ecosystems approach. *Progress in Physical Geography* 35: 671–680.

Fisher, B. and Turner, R.K., 2008. Ecosystem services: classification for valuation. *Biological Conservation* 141: 1167–1169.

Fisher, B., Costanza, R., Turner, R. *et al*., 2007. *Defining and Classifying Ecosystem Services for Decision Making.* CSERGE Working Paper EDM, No. 07-04. Available online: http://www.econstor.eu/bitstream/10419/80264/1/571829937.pdf (accessed 29 February 2016).

Fisher, B., Turner, R.K. and Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics* 68: 643–653.

Folke, C., 2003. Freshwater for resilience: a shift in thinking. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* 358: 2027–2036.

Folke, C., Hahn, T., Olsson, P. *et al*., 2005. Adaptive governance of social-ecological systems. *Annual Review of Environment and Resources* 30: 441–473.

Folke, C., Carpenter, S.R., Walker, B. *et al*., 2010. Resilience thinking: integrating resilience, adaptability and transformability. *Ecology and Society* 15: 20.

Fontana, V., Radtke, A., Fedrigotti, V.B. *et al*., 2013. Comparing land-use alternatives: using the ecosystem services concept to define a multi-criteria decision analysis. *Ecological Economics* 93: 128–136.

Forum for the Future, 2015. *The Five Capitals Model – A Framework for Sustainability*. Available online: https://www.forumforthefuture.org/sites/default/files/project/downloads/five-capitals-model.pdf (accessed 1 August 2016).

Garcia, X. and Pargament, D., 2015. Rehabilitating rivers and enhancing ecosystem services in a water-scarcity context: the Yarqon River. *International Journal Water Resources Development* 31: 1–15.

García-Nieto, A.P., García-Llorente, M., Iniesta-Arandia, I. *et al*., 2013. Mapping forest ecosystem services: from providing units to beneficiaries. *Ecosystem Services* 4: 126–138.

Gari, S.R., Newton, A. and Icely, J.D., 2015. A review of the application and evolution of the DPSIR framework with an emphasis on coastal social-ecological systems. *Ocean and Coastal Management* 103: 63–77.

Gaston, K.J., 1994. *Rarity*. Chapman and Hall, London, UK.

Gaston, K.J. and Williams, P.H., 1993. Mapping the world's species – the higher taxon approach. *Biodiversity Letters* 1: 2–8.

Georgiou, S., Bateman, I., Cole, M. *et al*., 2000. *Contingent Ranking and Valuation of River Water Quality Improvements: Testing for Scope Sensitivity, Ordering and Distance Decay Effects.* CSERGE Working Paper GEC 2000-18. Available online: http://cserge.ac.uk/sites/default/files/gec_2000_18.pdf (accessed 2 December 2015).

Ghermandi, A. and Fichtman, E., 2015. Cultural ecosystem services of multifunctional constructed treatment wetlands and waste stabilization ponds: time to enter the mainstream? *Ecological Engineering* 84: 615–623.

Giller, P.S., Covich, A.P., Ewel, K.C. *et al*., 2004. Vulnerability and management of ecological services in freshwater systems. In Wall, D.H. (ed.), *Sustaining Biodiversity and Ecosystem Services in Soils and Sediments*. Island Press, Washington DC, 275 pp.

Glenk, K., Lago, M. and Moran, D., 2011. Public preferences for water quality improvements: implications for the implementation of the EC Water Framework Directive in Scotland. *Water Policy* 13: 645–662.

Gobster, P.H., Nassauer, J.I., Daniel, T.C. *et al*., 2007. The shared landscape: what does aesthetics have to do with ecology? *Landscape Ecology* 22: 959–972.

Gordon, I.J., Martin-Ortega, J. and Ferrier, R.C., 2015. Introduction. In Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. and Khan, S. (eds). *Water Ecosystem Services: A Global Perspective*. Cambridge University Press, Cambridge, UK.

GWP (Global Water Partnership), 2000. *Integrated Water Resources Management*. GWP, Stockholm, Sweden. Available online: http://www.gwp.org/global/GWP-CACENA_files/en/pdf/tec04.pdf (accessed 26 February 2016).

Haines-Young, R. and Potschin, M., 2008. *England's Terrestrial Ecosystem Services and the Rationale for an Ecosystem Approach.* Full technical report to Defra (Project Code NR0107). Available online: http://www.nottingham.ac.uk/cem/pdf/NR107_FTR_080108.pdf (accessed 2 June 2016).

Haines-Young, R.H. and Potschin, M.B., 2009. *Methodologies for Defining and Assessing Ecosystem Services*. Final Report for JNCC (Project Code C08-0170-0062). Available online: http://www.nottingham.ac.uk/cem/pdf/JNCC_Review_Final_051109.pdf (accessed 1 June 2016).

Haines-Young, R. and Potschin, M.B., 2010a. *Proposal for a Common International Classification of Ecosystem Goods and Services (CICES) for Integrated Environmental and Economic Accounting*. Report to the European Environment Agency, Contract No. EEA/BSS/07/007. Available online: http://www.nottingham.ac.uk/cem/pdf/UNCEEA-5-7-Bk1.pdf (accessed 1 June 2016).

Haines-Young, R. and Potschin, M., 2010b. The links between biodiversity, ecosystem services and human well-being. In Raffaelli, D.G. and Frid, C.L.J. (eds), *Ecosystem Ecology: A New Synthesis*. Cambridge University Press, Cambridge, UK.

Haines-Young, R. and Potschin, M., 2013. The Common International Classification of Ecosystem Services. Consultation on Version 4, August–December 2012. *Report to the European Environment Agency*, Contract No EEA/IEA/09/003. Available online: http://www.nottingham.ac.uk/cem/pdf/CICES%20V43_Revised%20Final_Report_29012013.pdf (accessed 2 December 2015).

Haines-Young, R., Potschin, M., Fish, R. *et al*., 2007. *The Ecosystem Concept and the Identification of Ecosystem Goods and Services in the English Policy Context – A Review Paper*. DEFRA, London, UK.

Hairston, N.G., 1993. Cause–effect relationships in energy-flow, trophic structure, and interspecific interactions. *The American Naturalist* 142: 379–411.

Hanley, N., Schlapfer F. and Spurgeon, J., 2003. Aggregating the benefits of environmental improvements: distance decay functions for use and non-use values. *Journal of Environmental Management* 66: 297–304.

Hanley, N., Adamowicz, W. and Wright, R.E., 2005. Price vector effects in choice experiments: an empirical test. *Resource and Energy Economics* 27: 227–234.

Hanley, N., Wright, R.E. and Alvarez-Farizo, B., 2006. Estimating the economic value of improvements in river ecology using choice experiments: an application to the Water Framework Directive. *Journal of Environmental Management* 78: 183–193.

Harrington, R., Anton, C., Dawson, T.P. *et al*., 2010. Ecosystem services and biodiversity conservation: concepts and a glossary. *Biodiversity and Conservation* 19: 2778–2790.

Harrison, P.A., Berry, P.M., Simpson, G. *et al*., 2014. Linkages between biodiversity attributes and ecosystem services: a systematic review. *Ecosystem Services* 9: 191–203.

Haslett, J.R., Berry, P.M., Bela, G., *et al*., 2010. Changing conservation strategies in Europe: a framework integrating ecosystem services and dynamics. *Biological Conservation* 19: 2963–2977.

Hattam, C., Atkins, J.P., Beaumont, N. *et al*., 2015. Marine ecosystem services: linking indicators to their classification. *Ecological Indicators* 49: 61–75.

Hauck, J., Gorg, C., Varjopuro, R. *et al*., 2013. Benefits and limitations of the ecosystem services concept in environmental policy and decision making: some stakeholder perspectives. *Environmental Science and Policy* 25: 13–21.

Hauck, J., Stein, C., Schiffer, E. *et al*., 2015. Seeing the forest and the trees: facilitating participatory network planning in environmental governance. *Global Environmental Change* 35: 400–410.

Heberling, M., Fisher, A. and Shortle, J., 2000. *How the Number of Choice Sets Affects Responses in Stated Choice Surveys.* United States Environmental Protection Agency (US EPA), Cincinnati, OH.

Hein, L., van Koppen, K., de Groot, R.S. *et al*., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics* 57: 209–228.

Hering, D., Carvalho, L., Argillier, C. *et al*., 2015. Managing aquatic ecosystems and water resources under multiple stress – an introduction to the MARS project. *Science of the Total Environment* 503/504: 10–21.

Hirschi, C., Widmer, A., Briner, S. *et al*., 2013. Combining policy network and model-based scenario analyses: an assessment of future ecosystem goods and services in Swiss mountain regions. *Ecology and Society* 18: 42.

Hitzhusen, F.J., Ayalasomayajula, R. and Lowder, S., 2007. Economic analysis of infrastructure and water quality improvements in the Muskingum River corridor. In Hitzhusen, F.J. (eds) *Economic Valuation of River Systems*. Edward Elgard, Cheltenham, UK.

Hoff, H., 2011. *Understanding the Nexus*: *Background Paper for the Bonn 2011 Nexus Conference*. Stockholm Environment Institute, Stockholm. Available online: http://wef-conference.gwsp.org/fileadmin/documents_news/understanding_the_nexus.pdf (accessed 26 February 2016).

Holt, A.R., Moug, P. and Lerner, D.N., 2012. The network governance of urban river corridors. *Ecology and Society* 17: 25.

Hooper, D.U., Adair, E.C., Cardinale, B.J. *et al*., 2012. A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature* 486: 105–108.

Howe, C., Suich, H., Vira, B. *et al*., 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: a meta-analysis of ecosystem service trade-offs and synergies in the real world. *Global Environmental Change* 28: 263–275.

Huesker, F. and Moss, T., 2015. The politics of multi-scalar action in river basin management: implementing the EU Water Framework Directive (WFD). *Land Use Policy* 42: 38–47.

Hynes, S. and Hanley, N. 2006. Preservation versus development on Irish rivers: whitewater kayaking and hydro-power in Ireland. *Land Use Policy* 23: 170–180.

Hynes, S., Hanley, N. and Garvey, E., 2007. Up the proverbial creek without a paddle: accounting for variable participant skill levels in recreational demand modelling. *Environmental and Natural Resource Economics* 36: 413–426.

Hynes, S., Norton, D. and Hanley, N., 2013. Accounting for cultural dimensions in international benefit transfer. *Environmental and Resource Economics* 56: 499–519.

IIRC (International Integrated Reporting Council), 2013. *Capitals: Background Paper for <IR>*. Available online: http://integratedreporting.org/wp-content/uploads/2013/03/IR-Background-Paper-Capitals.pdf (accessed 1 August 2016).

Jackson, L.J., 2003. Macrophyte-dominated and turbid states of shallow lakes: evidence from Alberta lakes. *Ecosystems* 6: 213–223.

Jenkins, W.A., Murray, B.C., Kramer, R.A. *et al*., 2010. Valuing ecosystem services from wetlands restoration in the Mippissippi Alluvial Valley. *Ecological Economics* 69: 1051–1061.

Jobbagy, E. and Jackson, R., 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. *Ecological Applications* 10: 423–436.

Johnson, B.L., Richardson, W.B. and Naimo. T.J., 1995. Past, present, and future concepts in large river ecology. *BioScience* 45: 134–141.

Johnstone, C. and Markandya, A., 2006. Valuing river characteristics using combined site choice and participation travel cost models. *Journal of Environmental Management* 80: 237–247.

Jones, J.I. and Sayer, C.D., 2003. Does the fish-invertebrate-periphyton cascade precipitate plant loss in shallow lakes? *Ecology* 84: 2155–2167.

JRC (Joint Research Centre), 2012. *Indicators for Mapping Ecosystem Services: A Review*. Report EUR 25456 EN. Publications Office of the European Union, Luxembourg.

Kavanagh, P., Walsh, R. and Walsh, A., 2006. *Further Characterisation of Small Streams and Development of a New Small Stream Risk Score (SSRS) Project Output Report*. ESB International Limited, Ireland, 46 pp.

Kelble, C.R., Loomis, D.K., Lovelace, S. *et al*., 2013. The EBM-DPSER Conceptual Model: integrating ecosystem services into the DPSIR Framework. *PLoS ONE* 8: e70766.

Kelly, M., 2013. Data rich, information poor? Phytobenthos assessment and the Water Framework Directive. *European Journal of Phycology* 48: 437–450.

Kelly-Quinn, M., Blacklocke, S., Bruen, M. *et al*., 2014. Dublin Ireland: a city addressing challenging water supply, management, and governance issues. *Ecology and Society* 19: 10.

Kenter, J.O., Hyde, T., Christie, M. *et al*., 2011. The importance of deliberation in valuing ecosystem services in developing countries – evidence from the Solomon Islands. *Global Environmental Change and Policy Dimensions* 21: 505–521.

Kenter, J.O., Reed, M.S., Irvine, K.N. *et al*., 2014. *Shared, Plural and Cultural Values of Ecosystems. UK National Ecosystem Assessment Follow-on Report*. UNEP-WCMC, Cambridge, UK.

Kenter, J.O., O'Brien, L., Hockley, N. *et al*., 2015. What are shared and social values of ecosystems? *Ecological Economics* 111: 86–99.

Klamer, A., 2003. A pragmatic view on values in economics. *Journal of Economic Methodology* 10: 191–212.

Kokko, H. and López-Sepulcre, A., 2007. The ecogenetic link between demography and evolution: can we bridge the gap between theory and data? *Ecological Letters* 10: 773–782.

Kremen, C., 2005. Managing ecosystem services: what do we need to know about their ecology. *Ecological Letters* 8: 468–479.

Landers, D.H. and Nahlik, A.M., 2013. *Final Ecosystem Goods and Services Classification System* (FEGS-CS). EPA/600/R-13/ORD-004914. United States Environmental Protection Agency, Office of Research and Development, Washington, DC.

Lautenbach, S., Maes, J., Kattwinkel, M. *et al*., 2012. Mapping water quality-related ecosystem services: concepts and applications for nitrogen retention and pesticide risk reduction. *International Journal of Biodiversity Science, Ecosystem Services and Management* 8: 35–49.

Lavenus, R. and Chazot, S., 2015. *Shall We Trust Local Stakeholders to Manage Groundwaters?* International Commission on Irrigation and Damage 26th Euro-Mediterranean Regional Conference and Workshops: Innovate to Improve Irrigation Performances, 12–15 October, Montpellier, France. Available online: http://icid2015.sciencesconf.org/75087/document (accessed 2 December 2015).

Le Coq, J.-F., Froger, G., Pesche, D. *et al*., 2015. Understanding the governance of the Payment for Environmental Services Programme in Costa Rica: a policy process perspective. *Ecosystem Services* 16: 253–265.

Lele, S., Springate-Baginski, O., Lakerveld, R. *et al*., 2013. Ecosystem services: origins, contributions, pitfalls and alternatives. *Conservation and Society* 11: 343–358.

Lewis, S.L. and Maslin, M.A., 2015. Defining the Anthropocene. *Nature* 519: 171–180.

Limburg, K.E., O'Neill, R.V., Costanza, R. *et al*., 2002. Complex systems and valuation. *Ecological Economics* 41: 409–420.

Limburg, K.E., Luzadis, V.A., Ramsey, M. *et al*., 2010. The good, the bad, and the algae: perceiving ecosystem services and disservices generated by zebra and quagga mussels. *Journal of Great Lakes Research* 36: 86–92.

Locatelli, B., Imbach, P., Vignola, R. *et al*., 2011. Ecosystem services and hydroelectricity in Central America: modelling service flows with fuzzy logic and expert knowledge. *Regional Environmental Change* 11: 393–404.

Loreau, M., 2010. Linking biodiversity and ecosystems: towards a unifying ecological theory. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* 365: 49–60.

Luck, G.W., Daily, G.C., and Ehrlich, P.R., 2003. Population diversity and ecosystem services. *Trends in Ecology and Evolution* 18: 331–336.

Luck, G.W., Harrington, R., Harrison, P.A. *et al*., 2009. Quantifying the contribution of organisms to the provision of ecosystem services. *Bioscience* 59: 223–235.

Lyytimäki, J. and Sipilä, M., 2009. Hopping on one leg – the challenge of ecosystem disservices for urban green management. *Urban Forestry and Urban Greening* 8: 309–315.

Mace, G.M., Norris, K. and Fitter, A.H., 2012. Biodiversity and ecosystem services: a multi-layered relationship. *Trends in Ecology and Evolution* 27: 19–26.

McGarrigle, M.L., Bowman, J.J., Clabby, K.J. *et al*., 2002. *Water Quality in Ireland 1998–2000*. Environmental Protection Agency, Johnstown Castle, Ireland.

McShane, T.O., Hirsch, P.D., Trung, T.C. *et al*., 2011. Hard choices: making trade-offs between biodiversity conservation and human well-being. *Biological Conservation* 144: 966–972.

Maes, J., Liquete, C., Teller, A. *et al*., 2016. An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosystem Services* 17: 14–23.

Magnussen, K., Hasler, B. and Zandersen, M., 2015. *Ecosystem Services in Nordic Freshwater Management*. Nordic Council of Ministers, Denmark. Available online: http://norden.diva-portal.org/smash/get/diva2:767624/FULLTEXT01.pdf (accessed 2 December 2015).

Magurran, A.E., 2004. *Measuring Biological Diversity*. Blackwell Publishing, Oxford, UK.

Maltby, E., 1986. *Waterlogged Wealth*. Earthscan, London, UK.

Maltby, E. and Ormerod, S., 2011. Freshwaters – open waters, wetlands and floodplains. In *UK National Ecosystem Assessment Technical Report*, edition 1. UNEP-WCMC, Cambridge, UK, 295–360.

Maltby, E., Hogan, D.V., Immirzi, C.P. *et al*., 1994. Building a new approach to the investigation and assessment of wetland ecosystem functioning. In Mitsch, W.J. (eds), *Global Wetlands: Old World and New*. Elsevier, Amsterdam, 637–658.

Martin-Ortega, J., Jorda-Capdevila, D., Glenk, K. *et al*., 2015. What defines ecosystem services-based approaches? In Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. and Khan, S. (eds), *Water Ecosystem Services: A Global Perspective*. Cambridge University Press, Cambridge, UK.

Mastrangelo, M.E., Weyland, F., Herrera, L.P. *et al*., 2015. Ecosystem services research in contrasting socio--ecological contexts of Argentina: critical assessment and future directions. *Ecosystem Services* 16: 63–73.

May, R.M., 1988. How many species are there on Earth? *Science* 241: 1441–1449.

MCOS (M.C. O'Sullivan and Co. Ltd), 2013. *Three Rivers Project: Water Quality Monitoring and Management*. Available online: http://www.epa.ie/licences/lic_eDMS/090151b28024963c.pdf (accessed 1 June 2016).

MEA (Millennium Ecosystem Assessment), 2005. *Millennium Ecosystem Assessment, Ecosystems and Human Well-being: A Framework for Assessment.* Island Press, Washington, DC.

Mendoza, G.A. and Prabhu, R., 2003. Qualitative multi-criteria approaches to assessing indicators of sustainable forest resource management. *Forest Ecology and Management* 174: 329–343.

Metcalfe, P.J., Baker, W., Andrews, K. *et al*., 2012. An assessment of the nonmarket benefits of the Water Framework Directive for households in England and Wales. *Water Resources Research* 48: 1–18.

Metzger, M.J., Rounsevell, M.D.A., Acosta-Michlik, L. *et al*., 2006. The vulnerability of ecosystem services to land use change. *Agriculture, Ecosystems and Environment* 114: 69–85.

Meysman, F.J.R., Middelburg, J.J. and Heip, C.H.R., 2006. Bioturbation: a fresh look at Darwin's last idea. *Trends in Ecology and* Evolution 21: 688–695.

Ministerio de Medio Ambiente, 2000. *Plan estrategico Espagnol para la conservacion y el uso racional de los humedales en el marco de los ecosistemas acuatiicos de que dependen*. Spanish Government, Madrid.

Mitchell, R.C. and Carson, R.T., 1989. *Using Surveys to Value Public Goods: The Contingent Valuation Method*. Resources for the Future, Washington, DC.

Moberg, F. and Folke, C., 1999. Ecological goods and services of coral reef ecosystems. *Ecological Economics* 29: 215–233.

Mooney, H., 2009. The ecosystem-service chain and the biological diversity crisis. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* 365: 31–39

Mooney, H., Larigauderie, A., Cesario, M. *et al*., 2009. Biodiversity, climate change, and ecosystem services. *Current Opinion and Environmental Sustainability* 1: 46–54.

Morris, J. and Camino, M., 2011. Economic assessment of freshwater, wetland and floodplain (FWF) ecosystem services. In *UK National Ecosystem Assessment*. Available online: http://uknea.unep-wcmc.org/LinkClick.aspx?fileticket=IVLEq%2BxAI%2BQ%3D&tabid=82 (accessed 2 December 2015).

Moss, T., 2012. Spatial fit, from panacea to practice: implementing the EU Water Framework Directive. *Ecology and Society* 17: 2.

Muller, F. and Burkhard, B., 2012. The indicator side of ecosystem services. *Ecosystem Services* 1: 26–30.

Murphy, G., Hynes, S., Doherty, E. *et al*., 2014. *Estimating the Value to Irish Society of Benefits Derived from Water-related Ecosystem Services: a Discrete Choice Approach*. EPA STRIVE Report Series No. 127. Available online: https://www.epa.ie/pubs/reports/research/water/EPA_%20Research_%20Report_127.pdf (accessed 1 June 2016).

Murphy, J.F., Bowker, J.D., McFarland, B. *et al*., 2013. A diagnostic biotic index for assessing acidity in sensitive streams in Britain. *Ecological Indicators* 24: 562–572.

Murray, J.M.H., Meadows, A. and Meadows, P.S., 2002. Biogeomorphological implications of microscale interactions between sediment geotechnics and marine benthos: a review. *Geomorphology* 47: 15–30.

Naeem, S., Duffy, J.E. and Zavaleta, E., 2012. The functions of biological diversity in an age of extinction. *Science* 336: 1401–1406.

Nahlik, A.M., Kentula, M.E., Fennessy, M.S. *et al*., 2012. Where is the consensus? A proposed foundation for moving ecosystem service concepts into practice. *Ecological Economics* 77: 27–35.

Nassauer, J.I., Kosek, S.E., and Corry, R.C., 2001. Meeting public expectations with ecological innovation in riparian landscapes. *Journal of the American Water Resources Association* 37: 1439–1443.

Nelson, E., Mendoza, G., Regetz, J. *et al*., 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment* 7: 4–11.

NERA (NERA Economic Consulting) and Accent (Accent Market Research), 2007. *The Benefits of Water Framework Directive Programmes of Measures in England and Wales*. Report to DEFRA, Rep. CRP 4b/c. DEFRA, London, UK.

Neumann, A.C., 1966. Observations on coastal erosion in Bermuda and measurements of boring rate of sponge Cliona Lampa. *Limnology and Oceanography* 11: 92–108.

Niasse, M. and Cherlet, J., 2015. Using ecosystem services based approaches in integrated water resources management: perspective from the developing world. In Martin-Ortega, J., Ferrier, R.C., Gordon, I.J. *et al*. (eds), *Water Ecosystem Services*. Cambridge University Press, Cambridge, UK.

Niemeijer, D. and de Groot, R., 2008. A conceptual framework for selecting environmental indicator sets. *Ecological Indicators* 8: 14–25.

Norberg, J., 1999. Linking nature's services to ecosystems: some general ecological concepts. *Ecological Economics* 19: 183–202.

NPWS (National Parks and Wildlife Services), 2011. *Actions for Biodiversity 2011–2016, Ireland's National Biodiversity Plan*. Available online: https://www.npws.ie/legislation/national-biodiversity-plan (accessed 1 June 2016).

NSAD (National Strategy for Angling Development), 2015. *National Strategy for Angling Development*. Report to Inland Fisheries Ireland, Citywest, Dublin. Available online: http://www.fisheriesireland.ie/Angling-Information/national-strategy-for-angling-development.html (accessed 10 December 2015).

O'Grady, M., 2009. *Brown Trout in Ireland*. Central Fisheries Board, Dublin, Ireland.

Ormerod, S.J., 2014. Rebalancing the philosophy of river conservation. *Aquatic Conservation: Marine and Freshwater Ecosystems* 24: 147–152.

Othman, J., Bennett, J. and Blamey, R., 2004. Environmental values and resource management options: a choice modelling experience in Malaysia. *Environment and Development Economics* 9: 803–824.

Padawangi, R., Turpin, E., Herlily *et al*., 2016. Mapping an alternative community river: the case of the Ciliwung. *Sustainable Cities and Society* 20: 147–157.

Parker, N., Medcalf, K. and Naumann, E.-K., 2015. *Provision of National Ecosystem and Ecosystem Services Map for Ireland for a Suite of Prioritised Services*. Report commissioned by National Parks and Wildlife Service of the Department of Arts, Heritage and the Gaeltacht, Ireland. Available online: http://www.npws.ie/sites/default/files/files/ES%20Ireland%20-%20Workshop1Report.pdf (accessed 2 December 2015).

Parsons, M., Thoms, M., Capon, T. *et al*., 2009. *Resilience and Thresholds in River Ecosystems*. Waterlines Report, National Water Commission, Canberra, Australia.

Patterson, J.J., Smith, C. and Bellamy, J., 2013. Understanding enabling capacities for managing the 'wicked problem' of nonpoint source water pollution in catchments: a conceptual framework. *Journal of Environmental Management* 128: 441–452.

Perkins, D.M., McKie, B.G., Malmqvist, B. *et al*., 2010. Environmental warming and biodiversity-ecosystem functioning in freshwater microcosms: partitioning the effects of species identity, richness and metabolism. *Advances in Ecological Research* 43: 177–209.

Perkins, D.M., Bailey, R.A., Dossena, M. *et al*., 2015. Higher biodiversity is required to sustain multiple ecosystem processes across temperature regimes. *Global Change Biology* 21: 396–406.

Pielou, E.C., 1969. *An Introduction to Mathematical Ecology*. John Wiley, New York, NY.

Poirier, J. and Fleuret, A., 2010. *Using the Choice Experiment Method for Valuing Improvements in Water Quality: A Simultaneous Application to Four Recreation Sites of a River Basin*. Paper presented at the 59th Conference of Association Française de Science Economique, 9–10. Available online: https://feem-projectnet.serversicuro.it/exiopol/userfiles/POIRER_paper%20exiopol.pdf (accessed 12 July 2016).

Polizzi, C., Simonetto, M., Barausse, A. *et al*., 2015. Is ecosystem restoration worth the effort? The rehabilitation of a Finnish river affects recreational ecosystem services. *Ecosystem Services* 14: 158–169.

Postel, S. and Richter, B., 2003. *Rivers for Life: Managing Water for People and Nature*. Island Press, London, UK.

Potschin, M. and Haines-Young, R., 2011. Introduction to the special issue. *Progress in Physical Geography* 35: 571–574.

Prather, C.M., Pelini, S.L., Laws, A. *et al*., 2013. Invertebrates, ecosystem services and climate change. *Biological Reviews* 88: 327–348.

Purvis, A. and Hector, A., 2000. Getting the measure of biodiversity. *Nature* 405: 212–219.

Rathwell, K.J. and Peterson, G.D., 2012. Connecting social networks with ecosystem services for watershed governance: a social-ecological network perspective highlights the critical role of bridging organizations. *Ecology and Society* 17: 24.

Reiss J., Bailey, R.A., Cássio, F. *et al*., 2010. Assessing the contribution of micro-organisms and macrofauna to biodiversity-ecosystem functioning relationships in freshwater microcosms. *Advances in Ecological Research* 43: 150–176.

Reyjol, Y., Argillier, C., Bonne, W. *et al*., 2014. Assessing the ecological status in the context of the European Water Framework Directive: where do we go now? *Science of the Total Environment* 497–498: 332–344.

Ringold, P.L., Boyd, J., Landers, D. *et al*., 2009. *Report from the Workshop on Indicators of Final Ecosystem Services for Streams*. EPA/600/R-09/137. Available online: http://goo.gl/5ZFU4G (accessed 11 November 2015).

Rockström, J., Falkenmark, M., Allan, T. *et al*., 2015. The unfolding water drama in the Anthropocene: towards a resilience-based perspective on water for global sustainability. *Ecohydrology* 7: 1249–1261.

Rodríguez, J.P., Beard, Jr., T.D., Bennett, E.M. *et al*., 2006. Trade-offs across space, time, and ecosystem services. *Ecology and Society* 11: 28.

Rogers, K. and Biggs, H., 1999. Integrating indicators, endpoints and value systems in strategic management of the rivers of the Kruger National Park. *Freshwater Biology* 41: 439–451.

Salles, J.-M., 2011. Valuing biodiversity and ecosystem services: why put economic values on nature? *Comptes Rendus Biologies* 334: 469–482.

Sandbrook, C.G. and Burgess, N.D., 2015. Letter to Editor: Biodiversity and ecosystem services: not all positive. *Ecosystem Services* 12: 29.

Scheffer, M., 1997. *The Ecology of Shallow Lakes*. Chapman and Hall, London, UK.

Scheffer, M., Hosper, S.H., Meijer, M.L. *et al*., 1993. Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution* 8: 275–279.

Schröter, D., Cramer, W., Leemans, R. *et al*., 2005. Ecosystem service supply and vulnerability to global change in Europe. *Science* 310: 1333–1337.

Seppelt, R., Fath, B., Burkharde, B. *et al*., 2012. Form follows function? Proposing a blueprint for ecosystem service assessments based on reviews and case studies. *Ecological Indicators* 21: 145–154.

Shannon, C.E. and Weaver, E.W., 1949. *The Mathematical Theory of Communication*. University of Illinois, Champaign, IL.

Simpson, E.H., 1949. Measurement of diversity. *Nature* 163: 688.

Stein, C., Ernstson, H. and Barron, J., 2011. A social network approach to analyzing water governance: the case of the Mkindo catchment, Tanzania. *Physics and Chemistry of the Earth, Parts A/B/C* 36: 1085–1092.

Stirling A., 2007. A general framework for analysing diversity in science, technology and society. *Journal of The Royal Society Interface* 4: 707–719.

Stithou, M., Hynes, S., Hanley, N. *et al*., 2012. Estimating the value of achieving "Good Ecological Status" in the Boyne River Catchment in Ireland using choice experiments. *The Economic and Social Review* 43: 397–422. Available online: http://www.esr.ie/vol43_3/04%20ESRI%2043-3%20Stithou.pdf (accessed 1 June 2016).

Suding, K.N. and Gross, K.L., 2006. The dynamic nature of ecological systems: multiple states and restoration trajectories. In Falk, D.A., Palmer, M.A. and Zedler, J.B. (eds), *Foundations of Restoration Ecology*. Island Press, Washington, DC.

Sukhdev, P., Wittmer, H. and Miller D., 2014. The Economics of Ecosystems and Biodiversity (TEEB): challenges and responses. In Helm, D. and Hepburn, C. (eds), *Nature in the Balance: The Economics of Biodiversity.* Oxford University Press, Oxford.

Sullivan, C.A., Meigh, J.R., Giacomello, A.M. *et al*., 2003. The Water Poverty Index: development and application at the community scale. *Natural Resources Forum* 27: 189–199.

Suter II, G.W., 2006. *Ecological Risk Assessment*, second edition. CRC Press, Boca Raton, FL.

Syrbe, R-U. and Walz, U., 2012. Spatial indicators for the assessment of ecosystem services: providing, benefiting and connecting areas and landscape metrics. *Ecological Indicators* 21: 80–88.

Tallis, H., Kareiva, P., Marvier, M. *et al*., 2008. An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences of the United States of America* 105: 9457–9564.

TDI (Tourism Development International), 2013. *Socio-economic Study of Recreational Angling in Ireland*. Report to Inland Fisheries Ireland, Citywest, Dublin. Available online: http://www.fisheriesireland.ie/Angling-Information/socio-economic-survey-of-recreational-anglers.html (accessed 10 December 2015).

TEEB, 2010a. *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations*. Earthscan, London, UK.

TEEB, 2010b. *The Economics of Ecosystems and Biodiversity for Local and Regional Policy Makers*. Progress Press, Malta.

Tilman, D. and Downing, J.A., 1994. Biodiversity and stability in grasslands. *Nature* 367: 363–365.

Toman, M., 1998. Special section: forum on valuation of ecosystem services: Why not to calculate the value of the world's ecosystem services and natural capital. *Ecological Economics* 25: 57–60.

Tomczyk, A.M., White, P.C.L. and Ewertowski, M.W., 2016. Effects of extreme natural events on the provision of ecosystem services in a mountain environment: the importance of trail design in delivering system resilience and ecosystem service co-benefits. *Journal of Environmental Management* 166: 156–167.

Truchy, A., Angeler, D.G., Sponseller, R.A. *et al*., 2015. Linking biodiversity, ecosystem functioning and services, and ecological resilience: towards an integrative framework for improved management. *Advances in Ecological Research* 53: 55–96.

Turner, R., and Daily, G., 2008. The ecosystem services framework and natural capital conservation. *Environmental and Resource Economics* 39: 25–35.

Turner, R.K., van den Bergh, C.J.M., Soderqvist, T. *et al*., 2000. Ecological–economic analysis of wetlands: scientific integration for management and policy. *Ecological Economics* 35: 7–23.

UK NEA (National Ecosystem Assessment), 2011a. *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. UNEP-WCMC, Cambridge, UK.

UK NEA (National Ecosystem Assessment), 2011b. *The UK National Ecosystem Assessment: Technical Report*. UNEP-WCMC, Cambridge, UK.

UK NEA (National Ecosystem Assessment), 2014. *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. UNEP-WCMC and LWEC, UK.

UN Water, 2014. *The United Nations World Water Development Report – Water and Energy: Volume 1*. Available online: http://unesdoc.unesco.org/images/0022/002257/225741e.pdf (accessed 26 February 2016).

van Oudenhoven, A.P.E., Petz, K., Alkemade, R. *et al*., 2012. Framework for systematic indicator selection to assess effects of land management on ecosystem services. *Ecological Indicators* 21: 110–122.

Vandewalle, M., Sykes, M.T., Harrison, P.A. *et al*., 2009. *Review Paper on Concepts of Dynamic Ecosystems and their Services*. Available online: http://www.rubicode.net/rubicode/RUBICODE_Review_on_Ecosystem_Services.pdf (accessed 20 April 2016).

Vermeulen, S. and Koziell, I., 2002. *Integrating Global And Local Values: a Review of Biodiversity Assessment*. International Institute for Environment and Development, London, UK.

Villamor, G.B., Palomo, I., López Santiago, C.A. *et al*., 2014. Assessing stakeholders' perceptions and values towards social-ecological systems using participatory methods. *Ecological Processes* 3: 22.

Vlachopoulou, M., Coughlin, D., Forrow, D. *et al*., 2014. The potential of using the Ecosystem Approach in the implementation of the EU Water Framework Directive. *Science of the Total Environment* 470–471: 684–694.

Walker, B. and Meyers, J.A., 2004. Thresholds in ecological and social–ecological systems: a developing database. *Ecology and Society* 9: 3.

Walker, B. and Salt, D., 2006. *Resilience Thinking: Sustaining Ecosystems and People in a Changing World*. Island Press, Washington, DC.

Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. *Biological Conservation* 139: 235–246.

Wallace, K., 2008. Ecosystem services: multiple classifications or confusion? *Biological Conservation* 141: 353–354.

Waylen, K.A., Hastings, E.J., Banks, E.A. *et al*., 2014. The need to disentangle key concepts from ecosystem-approach jargon. *Conservation Biology* 28: 1215–1224.

Wells, M.P. and McShane, T.O., 2004. Integrating protected area management with local needs and aspirations. *Ambio* 33: 513–519.

Westman, W.E., 1977. How much are nature's services worth? Measuring the social benefits of ecosystem functioning is both controversial and illuminating. *Science* 197: 960–964.

Whittington, D., Smith, V.K., Okorafor, A. *et al*., 1992. Giving respondents time to think in contingent valuation studies – a developing-country application. *Journal of Environmental Economics and Management* 22: 205–225.

Whittington, D., Choe, C. and Lauria, D., 1997. The effect of giving respondents 'time to think' on tests of scope: an experiment in Calamba, Philippines. In Kopp, R., Pommerehne, W.W. and Schwartz, N. (eds), *Determining the Value of Non-marketed Goods*. Kluwer Academic Publisher, Boston, USA.

Wilson, C.M. and Matthews, W.H., 1970. *Man's Impact on the Global Environment: Report of the Study of Critical Environmental Problems (SCEP)*. MIT Press, Cambridge, MA.

Wilson, D., Farrell, C.A., Fallon, D. *et al*., 2016. Multi-year greenhouse gas balances at a rewetted temperate peatland. *Global Change Biology* in press.

Woodward, G., 2009. Biodiversity, ecosystem functioning and food webs in fresh waters: assembling the jigsaw puzzle. *Freshwater Biology* 54: 2171–2187.

Woodward, G., Bonada, N, Feeley, H.B. *et al*., 2015. Resilience of a stream community to extreme climatic events and long-term recovery from a catastrophic flood. *Freshwater Biology* 60: 2497–2510.

Zaccarelli, N., Petrosillo, I. and Zurlini, G., 2010. Natural capital security/vulnerability related to disturbance in a panarchy of social-ecological landscapes. In Jørgensen, S.E., Xu, L. and Costanza, R. (eds), *Handbook of Ecological Indicators for Assessment of Ecosystem Health*, second edition. CRC Press, Boca Raton, FL.

Zedler, J.B. and Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources* 30: 39–74.

Zhao, J., Liu, Q., Lin, L. *et al*., 2013. Assessing the comprehensive restoration of an urban river: an integrated application of contingent valuation in Shanghai, China. *Science of the Total Environment* 458–460: 517–526.

Zhu, J., Zhou, Y., Wang, S. *et al*., 2015. Multicriteria decision analysis for monitoring ecosystem service function of the Three-River Headwaters region of the Qinghai-Tibet Plateau, China. *Environmental Monitoring and Assessment* 187: 355.

Glossary

Asset(s)	See “Natural capital”
Beneficiaries	The interests of individuals and organisations (e.g. households, associations, societies and companies) that “drive active or passive consumption and/or appreciation of ecosystem services resulting in an impact (positive or negative) on their welfare” (Harrington *et al.*, 2010; Nahlik *et al.*, 2012; Landers and Nahlik, 2013)
Benefit	In this context, this is used as a general term to denote the many ways that human well-being is enhanced through the processes and functions of ecosystems via ecosystem services, or something that directly impacts on the welfare of people, such as more or better drinking water or a more satisfying fishing/angling trip (Fisher and Turner, 2008; Mace *et al.*, 2012). Benefits may be economic, social or health related (UK NEA, 2011a). However, it must be noted that “services” are not “benefits” (Boyd and Banzhaf, 2007)
Biodiversity	This can also be described as “biological diversity” and is the variability among living organisms from all sources including terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and between ecosystems (CBD, 2004; TEEB, 2010b)
DPSIR (see Abbreviations)	The causal framework for describing the interactions between society and the environment
Driver	Any natural or human-induced factor that directly or indirectly causes a change in an ecosystem (MEA, 2005; Harrington *et al.*, 2010; TEEB, 2010b)
Ecological process	An interaction among organisms; ecological processes frequently regulate the dynamics of ecosystems and the structure and dynamics of biological communities (Mace *et al.*, 2012)
Ecological stability	An ecosystem is considered ecologically stable if it has the capability to return to its original (and possibly dynamic) state after a disturbance and does not experience a regime shift, or if it exhibits low temporal variability or does not change dramatically as a result of a disturbance (TEEB, 2010b). See also “Pressure” and “Regime shift”.
Ecosystem	A dynamic complex of plant, animal and microorganism communities and their non-living environment that interact as a functional unit (MEA, 2005; TEEB, 2010b)
Ecosystem approach	A strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way (CBD, 2004)
Ecosystem function	A subset of the interactions between ecosystem structure and processes that underpin the capacity of an ecosystem to provide goods and services (TEEB, 2010b)
Ecosystem goods	The things from ecosystems that people value through experience, use or consumption, whether that value is expressed in economic, social or personal terms. Note that the use of this term goes well beyond a narrow definition of goods simply as physical items bought and sold in markets, and includes products that have no market price (e.g. outdoor recreation) (UK NEA, 2011a)

Ecosystem processes	Changes in the stocks and/or flows of materials in an ecosystem, resulting from interactions among organisms and with their physicochemical environment (Mace *et al*., 2012)
Ecosystem services	These are the benefits that people obtain from ecosystems (MEA, 2005). They can also be defined as the activities or functions of an ecosystem that provide benefits (or occasionally disservices) to humanity and its economy (e.g. Boyd and Banzhaf, 2007; Mace *et al*., 2012).
Ecosystem services approach	See "Ecosystem services framework"
Ecosystem services framework	A way of intervening to manage a system, based on taking a systemic and participatory approach. Also sometimes referred to as the "ecosystem services approach" (Waylen *et al*., 2014)
Evolutionary process	A process leading to changes in gene frequencies in populations and, potentially, ultimately the appearance of new species or intra-specific taxa (Mace *et al*., 2012)
Final ecosystem service	An ecosystem service (whether natural, semi-natural or highly modified) that directly underpins or delivers a good to humanity and improves well-being. A fundamental characteristic is that they retain a connection to the underlying ecosystem functions, processes and structures that generate them (Boyd and Banzhaf, 2007; UK NEA, 2011a; Mace *et al*., 2012; Haines-Young and Potschin, 2013)
Flow	The transfer of materials in an ecosystem from stocks and between pools, forms or states (Mace *et al*., 2012)
Human well-being	See "Well-being"
Indicator	Information based on measured data used to represent a particular attribute, characteristic or property of a system (TEEB, 2010b)
Intermediate service	A service that is not directly consumed by people but supports or underpins the output of other services. See also "Ecological function" (Haines-Young and Potschin, 2009)
Natural capital	The economic metaphor for the limited stocks of physical and biological resources found within an ecosystem and the capacity to provide for ecosystem services (TEEB, 2010b; Mace *et al*., 2012)
Pressure	A stress or negative effect on the environment caused by human activities (e.g. excess organic pollution) or by natural events (e.g. drought)
Regime	The set of system states within a stable landscape/catchment (Folke *et al*., 2010)
Regime shift	A change in a system state from one regime or stable state to another (Folke *et al*., 2010)
Resilience (of an ecosystem)	The capacity of an ecosystem to recover from and tolerate a disturbance to its structure and function without collapsing or changing status (Walker and Salt, 2006; Folke *et al*., 2010; Harrington *et al*., 2010; TEEB, 2010b)
Service-providing areas	A catchment or defined area that includes the sum of biodiversity and its traits (i.e. the biotic components of the ecosystem) required to deliver a given ecosystem service, as well as the physical or abiotic ecosystem components (Vandewalle *et al*., 2009; Syrbe and Walz, 2012)
Stakeholder	A person, group or organisation that has a stake in or is affected by the outcome of a particular activity or policy (Harrington *et al*., 2010; TEEB, 2010b)

Stock	The amount of materials or components in a given pool, form or state in an ecosystem that provide for services (Mace *et al*., 2012)
Threshold	Boundaries in space and time in changeable systems that separate alternative stable states (i.e. dynamic regimes) which shift towards different attractors (e.g. Briske *et al*., 2010; Folke *et al*., 2010)
Trade-offs	Management choices that intentionally or otherwise change the type, magnitude and relative mix of services provided by ecosystems (TEEB, 2010b)
Value	The magnitude of the improvement in well-being delivered to humans through the provision of good(s) (Mace *et al*., 2012)
Well-being	This arises from adequate access to the basic materials for a good life, which are needed to sustain freedom of choice and action, health, good social relations, security, peace of mind and spiritual experience. A state of well-being is dependent on the aggregated output of ecosystem goods and benefits, the provision of which can change the status of well-being (TEEB, 2010b; Haines-Young and Potschin, 2013)

Note: Definitions may have been adapted or amalgamated from cited sources to provide further clarity.

Abbreviations

BESS	Biodiversity & Ecosystem Service Sustainability
CBD	Convention on Biological Diversity
CICES	Common International Classification of Ecosystem Services
CO_2	Carbon dioxide
DAFM	Department of Agriculture, Food and the Marine
DEFRA	Department for Environment, Food & Rural Affairs
DPSIR	Driving forces–Pressures–States–Impacts–Responses (see Glossary for more details)
DURESS	Diversity of Upland Rivers for Ecosystem Service Sustainability
EBM-DPSER	Ecosystem Based Management – Driver, Pressure, State, Ecosystem Service, Response
EC	European Commission
EEA	European Environment Agency
EPA	Environmental Protection Agency (Ireland)
ESP	Ecosystem service provider
EU	European Union
FD	Floods Directive
FEGS	Final ecosystem goods and services
FEGS-CS	Final Ecosystem Goods and Services Classification System
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
IWRM	Integrated water resource management
MAES	Mapping and Assessment of Ecosystems and their Services
MCDA	Multi-criteria decision analysis
MEA	Millennium Ecosystem Assessment
NAICS	North American Industry Classification System
NAPCS	North American Product Classification System
NESCS	National Ecosystem Services Classification System
NPWS	National Parks and Wildlife Service
NSAD	National Strategy for Angling Development
ONEMA	Office national de l'eau et des milieux aquatiques (French National Agency for Water and Aquatic Environments)
PES	Payment for ecosystem services
SAC	Special Area of Conservation
SEEA	System Experimental Ecosystem Accounting
SNA	Social network analysis
SPA	Special Protection Area
SPU	Service providing units
TCM	Travel cost method
TDI	Tourism Development International
TEEB	The Economics of Ecosystems and Biodiversity
WFD	Water Framework Directive
WTP	Willingness to pay
UK NEA	United Kingdom National Ecosystem Assessment
UN	United Nations
UNEP	United Nations Environment Programme
UN WWDR	United Nations World Water Development Report
US EPA	United States Environmental Protection Agency

Appendix 1 Summary of Valuation Studies

Location	Study	Site, country	Data collection technique	Method	Subjected group	Attributes	Results
Ireland	Stithou *et al.* (2012)	Boyne River Catchment, Ireland	Primary	Choice experiment; stated preference	The public	River life, water appearance, recreational activities and condition of river banks	The highest values were found for “Good” water clarity and appearance (€35), “Good” river life (€28) and access to recreational activities (€23)
	Murphy *et al.* (2014) and Doherty *et al.* (2014)	Irish water bodies	Primary	Choice experiment; stated preference	The public	Biodiversity (aquatic ecosystem health), aesthetic values (water clarity and smell), recreation values (access to recreationally attractive sites) and overall landscape condition (erosion, banks damage, etc.).	The attributes most highly valued in this study were “Good” and “Moderate” water clarity and smell (€42 and €30 per person per year, respectively), and “Good” and “Moderate” health of ecosystems (€25 and €17, respectively). Lowest values were for “Access to recreational activities”
	Hynes and Hanley (2006)	Roughty River in Co. Kerry, Ireland	Primary	Travel cost; revealed preference	User group: whitewater kayakers	Water-based leisure activities	The mean individual consumer surplus was €235 per year. Aggregating this value to the country level, it resulted in a final estimate of €0.589 million spent per year by all Irish kayakers
	Curtis (2002)	Co. Donegal, Ireland	Primary	Travel cost; revealed preference	User group: anglers	Salmon angling	Angling quality, age and nationality were found to affect angling demand. The travel cost per day (including additional cost of accommodation and angling expenses) ranged from Irish £38.3 (€48.6) to £48.8 (€61.9) for Irish anglers
	Curtis (2003)	Ireland	Primary	Travel cost; Revealed preference	The public	Participation in water-based leisure activities (sea angling, boating, swimming and other beach/sea/ island day-trips)	Factors influencing participation varied across respondents. The cost of participation in water-based activities is a significant factor. More highly educated and/or female individuals and individuals on lower incomes are excluded from active participation

Location	Study	Site, country	Data collection technique	Method	Subjected group	Attributes	Results
International	Hitzhusen *et al*. (2007)	Muskingum River, Ohio, USA	Primary	Hedonic pricing; revealed preference	Zonal prices of houses	Water quality, bike paths establishment, locks and dams improvement and imposing zone and subdivision restrictions in surrounding counties	The value of the properties located in zoned areas is likely to increase by US $330 and $210 if the property has access to the river
	Metcalfe *et al*. (2012)	All of the water bodies (i.e. lakes, reservoirs, rivers, canals) in England and Wales	Primary	Contingent valuation (payment card), dichotomous choice and choice experiment; stated preference	The public	Water quality	The WTP results show local improvements are much more highly valued than national scale improvements. The mean WTP is between £44.5 per year and £167.9 per year. Improvements to high quality from medium quality are worth more than from low quality to medium quality
	Georgiou *et al*. (2000)	River Tame in Birmingham, England	Primary	Contingent ranking and contingent valuation; stated preference	The public	Improvements to biodiversity and recreation aspects	The results from contingent ranging are greater than the results from contingent valuation. The values for water quality were found for unit changes in various water quality indices. For example, £3–5 per household per annum for a unit increase in the Resources for the Future (RFF) Water Quality Index scale
	Birol *et al*. (2008)	Upper Silesia, Poland	Primary	Choice experiment, stated preference	The public	Two attributes referred to flooding (surface and underground), one to biodiversity abundance and one to recreational use	The results indicate that the average household is WTP the highest level for low flood risk, followed by river access and biodiversity (Polish zł14.5, zł4.6 and zł6.6, respectively)
	Glenk *et al*. (2011)	Rivers and lochs in Scotland	Primary	Choice experiment, stated preference	The public	Two differentiating factors: timescale and geographical location.	Respondents prefer improvement to current water quality status. Respondents are sensitive towards introduction of additional charge (water bill/tax). WTP estimates show that respondents' WTP varies between rivers and lochs
	Hanley *et al*. (2005)	River Wear (England) and River Clyde (Scotland)	Primary, Secondary	Choice experiment, Benefit transfer	The public	In-stream ecology (e.g. fish, invertebrates), aesthetic/ appearance (e.g. litter amount) and bankside conditions (e.g. vegetation/erosion	Highest values were found for maintaining bankside conditions (vegetation and erosion) and in-stream ecology (aquatic life: fish, plants, invertebrates), while aesthetic/appearance (amount of litter) had lower values. Values were significantly higher for the river Clyde The authors explored the applicability of the benefit transfer method. They found that despite many similarities, the test was rejected indicating that values cannot be transferred

AN GHNÍOMHAIREACHT UM CHAOMHNÚ COMHSHAOIL

Tá an Ghníomhaireacht um Chaomhnú Comhshaoil (GCC) freagrach as an gcomhshaol a chaomhnú agus a fheabhsú mar shócmhainn luachmhar do mhuintir na hÉireann. Táimid tiomanta do dhaoine agus don chomhshaol a chosaint ó éifeachtaí díobhálacha na radaíochta agus an truaillithe.

Is féidir obair na Gníomhaireachta a roinnt ina trí phríomhréimse:

Rialú: *Déanaimid córais éifeachtacha rialaithe agus comhlíonta comhshaoil a chur i bhfeidhm chun torthaí maithe comhshaoil a sholáthar agus chun díriú orthu siúd nach gcloíonn leis na córais sin.*

Eolas: *Soláthraímid sonraí, faisnéis agus measúnú comhshaoil atá ar ardchaighdeán, spriocdhírithe agus tráthúil chun bonn eolais a chur faoin gcinnteoireacht ar gach leibhéal.*

Tacaíocht: *Bímid ag saothrú i gcomhar le grúpaí eile chun tacú le comhshaol atá glan, táirgiúil agus cosanta go maith, agus le hiompar a chuirfidh le comhshaol inbhuanaithe.*

Ár bhFreagrachtaí

Ceadúnú

Déanaimid na gníomhaíochtaí seo a leanas a rialú ionas nach ndéanann siad dochar do shláinte an phobail ná don chomhshaol:

- saoráidí dramhaíola *(m.sh. láithreáin líonta talún, loisceoirí, stáisiúin aistrithe dramhaíola)*;
- gníomhaíochtaí tionsclaíocha ar scála mór *(m.sh. déantúsaíocht cógaisíochta, déantúsaíocht stroighne, stáisiúin chumhachta)*;
- an diantalmhaíocht *(m.sh. muca, éanlaith)*;
- úsáid shrianta agus scaoileadh rialaithe Orgánach Géinmhodhnaithe *(OGM)*;
- foinsí radaíochta ianúcháin *(m.sh. trealamh x-gha agus radaiteiripe, foinsí tionsclaíocha);*
- áiseanna móra stórála peitril;
- scardadh dramhuisce;
- gníomhaíochtaí dumpála ar farraige.

Forfheidhmiú Náisiúnta i leith Cúrsaí Comhshaoil

- Clár náisiúnta iniúchtaí agus cigireachtaí a dhéanamh gach bliain ar shaoráidí a bhfuil ceadúnas ón nGníomhaireacht acu.
- Maoirseacht a dhéanamh ar fhreagrachtaí cosanta comhshaoil na n-údarás áitiúil.
- Caighdeán an uisce óil, arna sholáthar ag soláthraithe uisce phoiblí, a mhaoirsiú.
- Obair le húdaráis áitiúla agus le gníomhaireachtaí eile chun dul i ngleic le coireanna comhshaoil trí chomhordú a dhéanamh ar líonra forfheidhmiúcháin náisiúnta, trí dhíriú ar chiontóirí, agus trí mhaoirsiú a dhéanamh ar leasúchán.
- Cur i bhfeidhm rialachán ar nós na Rialachán um Dhramhthrealamh Leictreach agus Leictreonach (DTLL), um Shrian ar Shubstaintí Guaiseacha agus na Rialachán um rialú ar shubstaintí a ídíonn an ciseal ózóin.
- An dlí a chur orthu siúd a bhriseann dlí an chomhshaoil agus a dhéanann dochar don chomhshaol.

Bainistíocht Uisce

- Monatóireacht agus tuairisciú a dhéanamh ar cháilíocht aibhneacha, lochanna, uiscí idirchriosacha agus cósta na hÉireann, agus screamhuiscí; leibhéil uisce agus sruthanna aibhneacha a thomhas.
- Comhordú náisiúnta agus maoirsiú a dhéanamh ar an gCreat-Treoir Uisce.
- Monatóireacht agus tuairisciú a dhéanamh ar Cháilíocht an Uisce Snámha.

Monatóireacht, Anailís agus Tuairisciú ar an gComhshaol

- Monatóireacht a dhéanamh ar cháilíocht an aeir agus Treoir an AE maidir le hAer Glan don Eoraip (CAFÉ) a chur chun feidhme.
- Tuairisciú neamhspleách le cabhrú le cinnteoireacht an rialtais náisiúnta agus na n-údarás áitiúil *(m.sh. tuairisciú tréimhsiúil ar staid Chomhshaol na hÉireann agus Tuarascálacha ar Tháscairí)*.

Rialú Astaíochtaí na nGás Ceaptha Teasa in Éirinn

- Fardail agus réamh-mheastacháin na hÉireann maidir le gáis cheaptha teasa a ullmhú.
- An Treoir maidir le Trádáil Astaíochtaí a chur chun feidhme i gcomhair breis agus 100 de na táirgeoirí dé-ocsaíde carbóin is mó in Éirinn.

Taighde agus Forbairt Comhshaoil

- Taighde comhshaoil a chistiú chun brúnna a shainaithint, bonn eolais a chur faoi bheartais, agus réitigh a sholáthar i réimsí na haeráide, an uisce agus na hinbhuanaitheachta.

Measúnacht Straitéiseach Timpeallachta

- Measúnacht a dhéanamh ar thionchar pleananna agus clár beartaithe ar an gcomhshaol in Éirinn *(m.sh. mórphleananna forbartha)*.

Cosaint Raideolaíoch

- Monatóireacht a dhéanamh ar leibhéil radaíochta, measúnacht a dhéanamh ar nochtadh mhuintir na hÉireann don radaíocht ianúcháin.
- Cabhrú le pleananna náisiúnta a fhorbairt le haghaidh éigeandálaí ag eascairt as taismí núicléacha.
- Monatóireacht a dhéanamh ar fhorbairtí thar lear a bhaineann le saoráidí núicléacha agus leis an tsábháilteacht raideolaíochta.
- Sainseirbhísí cosanta ar an radaíocht a sholáthar, nó maoirsiú a dhéanamh ar sholáthar na seirbhísí sin.

Treoir, Faisnéis Inrochtana agus Oideachas

- Comhairle agus treoir a chur ar fáil d'earnáil na tionsclaíochta agus don phobal maidir le hábhair a bhaineann le caomhnú an chomhshaoil agus leis an gcosaint raideolaíoch.
- Faisnéis thráthúil ar an gcomhshaol ar a bhfuil fáil éasca a chur ar fáil chun rannpháirtíocht an phobail a spreagadh sa chinnteoireacht i ndáil leis an gcomhshaol (*m.sh. Timpeall an Tí, léarscáileanna radóin).*
- Comhairle a chur ar fáil don Rialtas maidir le hábhair a bhaineann leis an tsábháilteacht raideolaíoch agus le cúrsaí práinnfhreagartha.
- Plean Náisiúnta Bainistíochta Dramhaíola Guaisí a fhorbairt chun dramhaíl ghuaiseach a chosc agus a bhainistiú.

Múscailt Feasachta agus Athrú Iompraíochta

- Feasacht chomhshaoil níos fearr a ghiniúint agus dul i bhfeidhm ar athrú iompraíochta dearfach trí thacú le gnóthais, le pobail agus le teaghlaigh a bheith níos éifeachtúla ar acmhainní.
- Tástáil le haghaidh radóin a chur chun cinn i dtithe agus in ionaid oibre, agus gníomhartha leasúcháin a spreagadh nuair is gá.

Bainistíocht agus struchtúr na Gníomhaireachta um Chaomhnú Comhshaoil

Tá an ghníomhaíocht á bainistiú ag Bord lánaimseartha, ar a bhfuil Ard-Stiúrthóir agus cúigear Stiúrthóirí. Déantar an obair ar fud cúig cinn d'Oifigí:

- An Oifig um Inmharthanacht Comhshaoil
- An Oifig Forfheidhmithe i leith cúrsaí Comhshaoil
- An Oifig um Fianaise is Measúnú
- An Oifig um Cosaint Raideolaíoch
- An Oifig Cumarsáide agus Seirbhísí Corparáideacha

Tá Coiste Comhairleach ag an nGníomhaireacht le cabhrú léi. Tá dáréag comhaltaí air agus tagann siad le chéile go rialta le plé a dhéanamh ar ábhair imní agus le comhairle a chur ar an mBord.

EPA Research Report 187

ESManage Literature Review Ecosystem Services in Freshwaters

Authors: Hugh B. Feeley, Michael Bruen, Craig Bullock, Mike Christie, Fiona Kelly, Kyriaki Remoundou, Ewa Siwicka & Mary Kelly-Quinn

epaResearch
Climate - Water - Sustainability
Identifying pressures • Informing policy • Developing solutions

Identify pressures

Ireland's freshwaters are among the best in Europe. However, they are under increasing pressure from a range of land-use and other anthropogenic pressures, especially from elevated nutrients (nitrogen and phosphorus) and sediment inputs. The continuing loss of high status waters is a key concern. Planned future land-use intensification for food production, together with climate change will further stress aquatic resources both in terms of quality and quantity. The ESManage Literature Review highlights how pressures have implications for a range of ecosystem services derived from freshwaters.

Inform policy

The ESManage Literature Review considers how the ecosystem services framework aligns with the objectives of current policy and legislation to inform management of freshwater resources. The Water Framework Directive (WFD) is the key EU driver requiring Member States to improve and sustainably manage water quality. The specific benefits of incorporating the ecosystem services framework into the implementation of the WFD relate to illustrating how human wellbeing is dependent on good ecological health and widening the focus from good ecological status as an end in itself to showing how it supports societal goals. Additionally, it allows for the proper assessment and communication of the benefits and co-benefits of implementing the WFD, highlighting potential trade-offs involved in selecting cost-effective measures but also avoiding unintended impacts of measures on other benefits (not directly associated with the measure). Other relevant policy measures such as the EU Biodiversity Strategy aim to halt the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restore them in so far as feasible.

Develop solutions

Identification of the chain of knowledge and data needs, as outlined in the ESManage Literature Review, is a key step in efforts to incorporate the ecosystem services framework into policy related to the management of freshwater resources. This review details these information needs and associated knowledge gaps, especially with respect to understanding the complex ecological linkages between the health and resilience of the ecosystem (critically dependent on biodiversity) and the provision of ecosystem services, converting this understanding into projections of possible future changes in ecosystem services provision that can be understood by the wider public, and identifying the means by which this public can value such changes to ecosystem service provision.

EPA Research: McCumiskey House,
Richiew, Clonskeagh, Dublin 14.

Phone: 01 268 0100
Twitter: @EPAResearchNews
Email: research@epa.ie

www.epa.ie

EPA Research Webpages
www.epa.ie/researchandeducation/research/