A Framework for Freshwater Ecosystem Management

Scientific Background for regional consultations on developing water quality guidelines for ecosystems

Volume 4

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United Nations Environment Programme
P.O. Box 47074
Nairobi, 00100, Kenya
Tel: (+254) 20 7621234
E-mail: unenvironment-water@un.org
Web: www.unenvironment.org/water

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A Framework for Freshwater Ecosystem Management

Volume 4: Scientific Background

Table of Contents

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This volume was prepared on behalf of the United Nations Environmental Programme (UN Environment) by the Editorial Team of the United Nations University - Institute for Environment and Human Security (UNU-EHS). This team was composed of Mr Janos Bogardi, Mr Fabrice Renaud, Ms Zita Sebesvari, Ms Nike Sommerwerk and Ms Yvonne Walz supported by Ms Aarti Basnyat, Ms Susanne Haas, Ms Janine Kandel, Ms Aileen Orate, Ms Mariko Shimazu and Ms Sijia Yi.

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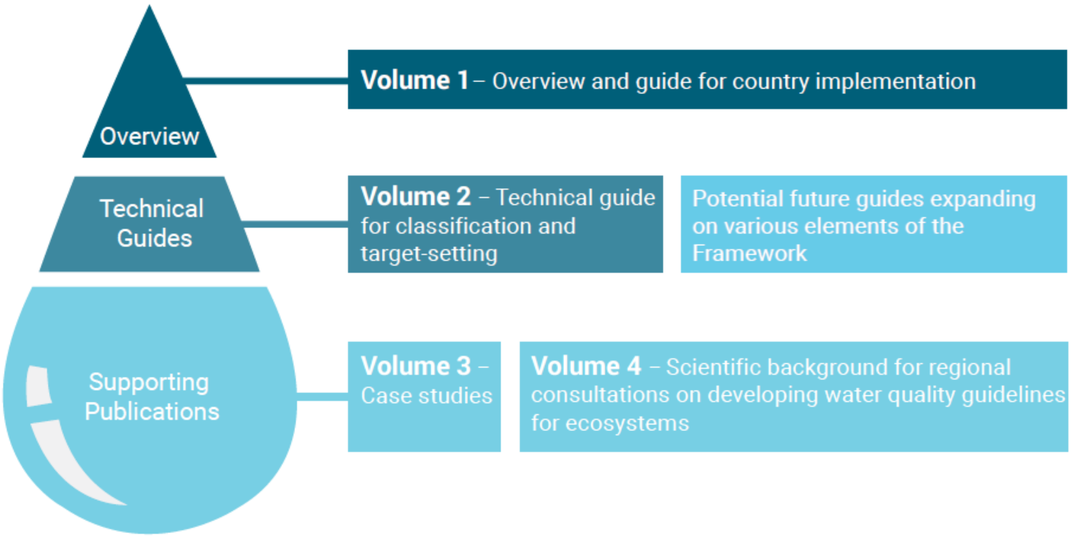
Preface: A Framework for Freshwater Ecosystem Management

The UN Environment 'Framework for Freshwater Ecosystem Management' series presents a holistic management framework to guide country-level action to sustainably manage freshwater ecosystems. It builds on the decision by the UN Environment Programme (UNEP) Governing Council to develop water quality guidelines for ecosystems (Decision 27/3, 2013).

The Framework supports national and international goals related to freshwater ecosystems, such as relevant Aichi Biodiversity Targets and Sustainable Development Goal (SDG) targets. An overview of the series, which currently consists of four volumes, is provided below:

The Freshwater Ecosystem Management series



Volume 1 provides an overview of the Framework, and places it in the context of supporting Agenda 2030. It is intended for a wide audience, including decision makers, practitioners, scientists, non-governmental organizations and the general public.

Volume 2 describes aspects of the Framework in more technical detail: classification systems for freshwater ecosystem types, setting targets for ecological status, and monitoring progress against these targets. It is primarily aimed at government agency staff responsible for the sustainable management of freshwater ecosystems. These aspects have been selected for elaboration as they are likely to be the most useful for the largest number of countries in relation to Aichi Biodiversity Targets and the SDGs. Additional technical guides that expand on other parts of the Framework, such as the design and implementation of remediation actions, may be developed depending on demand from countries.

Volume 3 – Case Studies – provides examples from around the world, illustrating different aspects of the Framework.

Volume 4 – Scientific Background – underpins the series and includes a review of water quality guidelines for ecosystems from around the world. It was the first volume in the series to be developed, beginning in 2013 over a number of years.

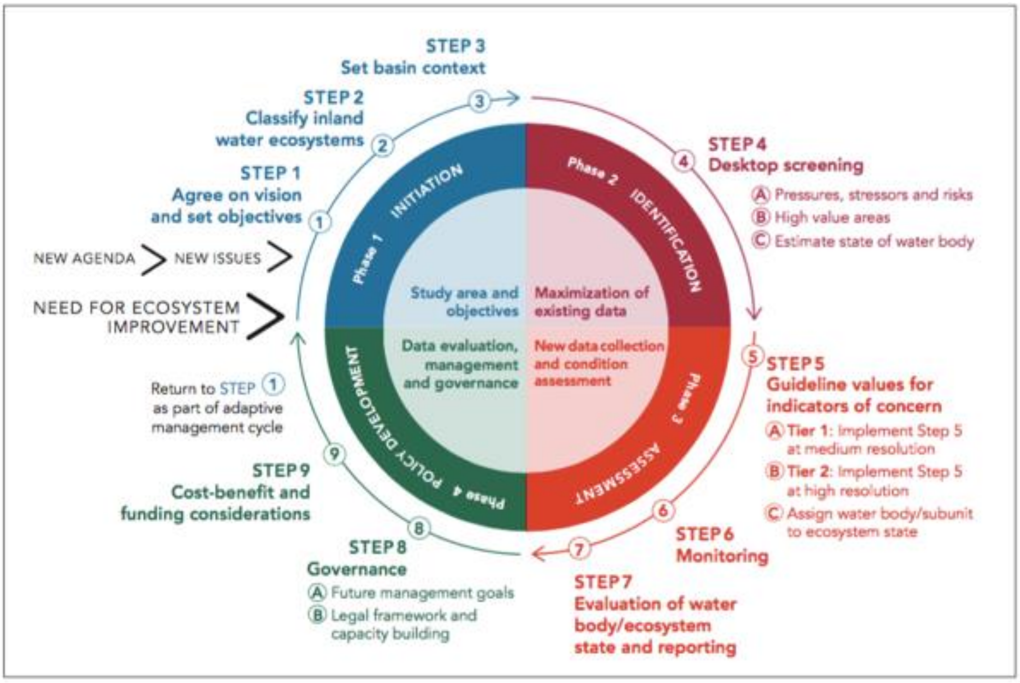
Volume 4: Scientific Background

Following a UN Environment Governing Council decision in 2013 to create what was then referred to as International Water Quality Guidelines for Ecosystems (IWQGES), work on this Scientific Background volume commenced in May 2013 and culminated in work to support an advanced draft in March 2016.[1] The drafting process was led by the United Nations University - Institute for Environment and Human Security (UNU-EHS), implemented by a Drafting Group made up of ten global experts, and guided by an Advisory Group consisting of representatives from 19 countries (see Acknowledgements section in this volume). Due to the significant contributions from experts and country representatives over several years to the Scientific Background, this volume is presented largely unchanged from the advanced draft for regional consultations finalised in March 2016. Minor changes have been made to update the Introduction (Chapter 1) and Conclusions (Chapter 6), as well as to adapt the language in the report to reflect the revised focus on the management framework, as opposed to water quality guidelines. Other than that, the scientific and technical content in Chapters 2 – 5 remain unchanged.

Differences in the Framework in this volume and the rest of the series

The advanced draft was reviewed in regional consultations held between May 2016 and February 2017. Following this, Volumes 1 and 2 were developed to address feedback received during the regional consultations, as well as to consider the further development of targets, indicators and reporting processes within the 2030 Agenda for Sustainable Development and its set of Sustainable Development Goals. This led to a clarification of the focus and objectives of the Freshwater Ecosystems Management Framework, and some minor revisions to the phases and steps within it. A full history of the development of the Framework is presented in Volume 1, Annex 1. [2]

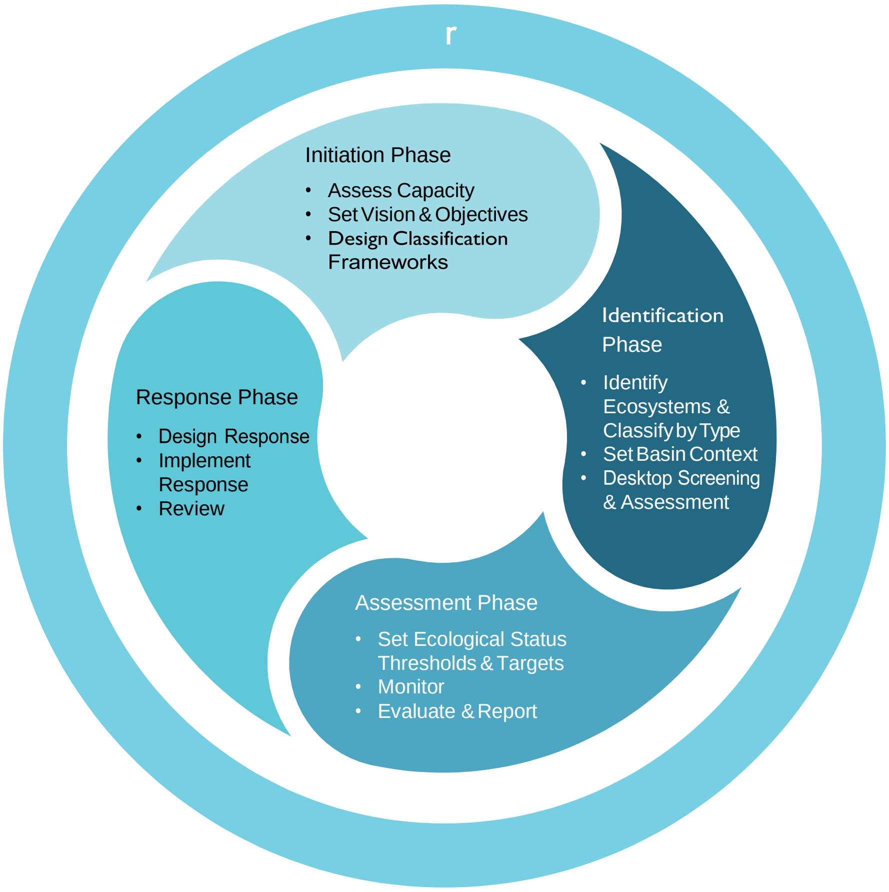
Figure 0-1 Advanced Draft Framework for Regional Consultations (March 2016). 4 Phases, 9 Steps.



1 Available at http://www.unep.org/sites/default/files/Documents/20160315_iwqges_pd_final.pdf

2 UN Environment 2017. Framework for Freshwater Ecosystem Management: Volume 1 – An overview and guide towards implementing Agenda 2030.

Figure 0-2 Final Framework for Freshwater Ecosystem Management (December 2017). 4 Phases, 12 Steps, with outer governance band.



The main change in the way the Framework is presented has been a shift in focus from the previously used 'International Water Quality Guidelines for Ecosystems', to the 'Framework for Freshwater Ecosystem Management'. This change, based on feedback from stakeholders during the regional consultation period, was made to better reflect the core focus and purpose of the work. The Framework is broader than water quality guidelines, though these are still addressed within the Framework.

The main differences between the Framework in the advanced draft (March 2016) developed for Regional Consultations, and the final Framework in December 2017, are shown in the table below.

The Response Phase was added to the Framework to make it more of a complete management Framework. The previous 'Policy Development' Phase was transformed into the outer governance circle, recognizing that governance aspects would be ongoing, and can impact on, and be guided by, any aspect of the Framework. The step numbers were removed to emphasize that many of the steps may be undertaken concurrently, even though they are broadly set out in a logical order.

	2017 Framework	2016 Draft Framework for Regional Consultations
Phase	**Steps**	**Steps**
Initiation	**Assess capacity**	Not included in draft Framework.
	Set vision/objectives	1. Agree on vision and set objectives
	Design Classification frameworks	2. Classify inland water ecosystems
Identification	**Identify Ecosystems and Classify by Type**	
	Set basin context	3. Set basin context (Phase 1)
	Desktop Screening	4. Desktop Screening (Phase 2)
Monitoring	**Set Ecological Status Thresholds and Targets**	5. Guideline values for indicators of concern
	Monitoring	6. Monitoring
	Evaluate & report	7. Evaluation of waterbody/ecosystem state and reporting
Response	**Design response**	Response Phase not included in draft Framework. Instead, Phase 4 was Policy Development, and included step 8. Governance and 9. Cost-benefit and funding considerations. These issues mainly addressed through the new Capacity Assessment step in the Initiation Phase, and the new 'Governance' outer circle.
	Implement response	
	Review	

Glossary

Acidification: The lowering of pH in soils or water. Commonly associated with changes caused by external processes such as acid precipitation and acidic runoff.

Adaptive management: The mode of management in which an intervention (action) is followed by monitoring (learning), with the information then is being used in designing and implementing the next intervention (acting again) to steer the system towards a given objective or to modify the objective itself (MEA, 2003).

Aerobic: Organisms requiring oxygen for respiration or conditions where oxygen is available.

Alluvial: Unconsolidated soil transported and deposited by water. In this context alluvial channels refer to cases where water flows in a bed formed from its own sediment.

Anions: Negatively charged ions.

Aquatic ecosystems: For the purposes of developing International Water Quality Guidelines for Ecosystems, aquatic ecosystems are defined as the abiotic (physical and chemical) and biotic components, habitats and ecological processes contained within rivers and their riparian zones, reservoirs, lakes and wetlands and their fringing vegetation. For the purposes of these guidelines, the scope has been restricted to inland surface and transitional waters (deltas, estuaries), excluding marine and estuarine systems as well as groundwater.

Aquifer: A water-bearing layer of soil, sand, gravel or rock that will yield usable quantities of water to a well.

Base flow: Sustained, low flow in a stream; groundwater discharge is the source of base flow in most places.

Benchmark: A benchmark is a chemical concentration or any similar attribute, specific to either water or sediment, to indicate a certain harm or risk to humans or animals in the environment. Benchmarks are meant to be used for screening purposes only; they are not regulatory criteria (US EPA) or standards (EU WFD). Benchmarks also serve to identify of certain desirable levels (either to be achieved or to not fall below).

Benthic: Inhabiting the bottom of a water body.

Benthos: Plants and animals that live on, in, or attached to the stream, river, lake or sea bottom.

Best Attainable Condition (BAC): Best attainable condition is equivalent to the expected ecological condition of least-disturbed sites if the best possible management practices were in use for some period of time (Stoddard *et al.*, 2006).

Biochemical oxygen demand (BOD): the amount of dissolved oxygen (mg/l) which would be consumed by organisms in water rich in organic matter to decompose those.

Biodiversity: The variety of life in all its forms contained within a given space at a particular time.

Bioindicator: An organism and/or biological process whose change in numbers, structure, or function points to changes in the integrity or quality of the environment.

Biological criteria or biocriteria: Narrative expressions or numeric values of the biological characteristics of aquatic communities based on appropriate reference conditions; as such, biological criteria serve as an index of aquatic community health.

Biome: A large naturally occurring community of flora and fauna occupying a major habitat.

Brackish water: Salty water (> 0.5‰ salt) with less salt than seawater.

Buffer strip: The vegetation along a stream left intact after logging or land clearing. An intact buffer strip (also called riparian zone) prevents fine sediment from entering into a stream.

Buffering capacity: A measure of the relative sensitivity of a solution to pH changes on addition of acids or bases.

Calcareous: Containing salts of calcium, for example, calcium carbonate as limestone rock or derived soil.

Carcinogenic: Capable of causing, promoting or giving rise to the development of cancer.

Cation: An ion with a positive charge.

Catchment basin: The land area that contributes water to a stream or lake system or directly to the ocean; also referred to as a drainage basin.

Cenotes: Natural pits or sinkholes resulting from the collapse of limestone bedrock that exposes groundwater underneath.

Chemical oxygen demand (COD): The amount of oxygen required to oxidise all the organic matter that is susceptible to oxidation by a strong chemical oxidant.

Chlorophyll (-a, -b, -c): The green pigment in plants and algae that during photosynthesis captures sunlight energy and converts it into chemical energy in the form of carbohydrates. Chlorophyll-a, -b and -c are forms of chlorophyll found in different proportions in different plants. Chlorophyll is used as a measure of the quantity of algae in water.

Clarity: Refers to the depth to which light can penetrate in a water body and is measured by the depth to which a Secchi disk (a 20 cm diameter disk printed with black and white quadrants) is visible.

Classification: Any set of formal categories into which a particular field of data is partitioned. See related term: Typology.

Criteria: Elements of state water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.

Cryptic species: Two or more distinct species that are erroneously classified (and hidden) under one species name.

Diffuse (non-point) pollutant sources: Diffuse discharges of pollutants from surface run off, infiltration or atmospheric sources.

Driver: Any natural or human-induced factor that directly or indirectly causes a change in an ecosystem.

Ecocentric: Addressing questions related to ecosystems without considering societal demands and human use.

Ecospecs and userspecs: In managing water quality, as in all management, there is a need to set objectives. Objectives for different levels of ecosystem health are termed ecospecs (ecological specifications). Objectives are also to meet the needs of different users (userspecs). When ecological and user-requirement objectives are integrated they are termed Resource Quality Objectives (RQOs) (Palmer *et al.,* 2004b).

Ecosystem: A dynamic complex of plant, animal and microorganism communities and their non-living environment interacting as a functional unit.

Ecosystem health (concept): A measure of the stability and sustainability of ecosystem functioning or ecosystem services that depend on an ecosystem being active and maintaining its organization, autonomy and resilience over time. Ecosystem health contributes to human well-being through sustainable ecosystem services and conditions for human health.

Ecosystem services: The benefits people obtain from ecosystems. These include provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling that maintain the conditions for life on Earth (MEA*,* 2003).

Endemic species: Species native to or limited to a certain region.

Endocrine disruptors: Chemicals which interfere with the endocrine (or hormone system) in animals and humans. These disruptions can cause cancerous tumours, birth defects, gender disorientation and other development disorders. Any system in the body controlled by hormones can be derailed by hormone disruptors.

Environmental flow (e-flow): Quantity, timing, and quality of river water flows required to maintain specified, valued features of the ecosystem (Tharme and King, 1998).

Ephemeral: Short duration phenomena with long periods between subsequent occurrences (such as short streamflow after heavy downpours in arid regions).

Eutrophication: Enhanced primary productivity caused by nitrogen and phosphorus, organic pollution, intense catchment land use and habitat degradation affect almost all European surface waters. Ecosystem functions have been lost and many aquatic species have disappeared from entire ecoregions.

Floodplain: The land bordering a stream, built up of sediments from stream overflow and subject to inundation when the stream floods.

Fractionation: Separation; break up of a mixture into different portions / components especially by a fractional process based on differences in physical or chemical properties.

Groundwater: Water in the zone of saturation where all open spaces in sediment and rock are completely filled with water.

Guideline(s): 'Guideline(s)' refers to a document conceived and compiled as a possible roadmap and recommendations helping the respective government authorities to establish (potentially binding) standards for water quality requirements for ecosystem health within their respective jurisdiction. Also used as obligatory or recommended prescriptions of water (quality) attributes for the achievement of certain water (quality) states.

Habitat: Habitat is the place where a particular species lives and grows. It is the essential environment - at least the physical environment - that surrounds (influences and is utilized by) a species population.

Hydrogeomorphic Index: Based on their geomorphic position and hydrologic characteristics, seven different wetland classes were identified by Brinson (1993): Depressional wetlands, riverine wetlands, mineral flats, organic flats, tidal flats, lacustrine fringe and slope wetlands.

Hydromorphology: The physical characteristics of the shape, boundaries and content of a water body. The hydromorphological quality elements for classification of ecological status are listed in Annex V of the EU WFD.

Impact: The effect of a stressor on the environment (e.g. fish kill, algal bloom, extinction, ecosystem modification).

Index (plural: Indices): An aggregate of indicators. Indices can be derived from individual indicators by following certain aggregation rule(s) to form an index. An index may be composed of indicators having different dimensions. Indices are frequently dimensionless. Indices are composites and can be handled as semi-quantitative or qualitative measures. Indices are often used to assign specific water bodies or ecosystems to a certain quality category or for potential use.

Index of biological/biotic integrity (IBI): The Index of Biotic Integrity (IBI) was first developed by Karr (1981) to help resource managers sample, evaluate, and describe the condition of small warm water streams based on fish species. The IBI is a metric based on the concept that certain groups of aquatic organisms are sensitive to pollution while others are more tolerant. It allows a standardized comparison of sites along a stream, or multiple streams, across geographic areas and through time. The IBI is an integrative expression of site condition across multiple metrics. Nowadays many different modifications and applications of the IBI exist.

Indicator: An ecological indicator is defined here as a measure (directly measured or inferred) or a model that characterizes an ecosystem or one of its critical components. An indicator may reflect biological, chemical or physical and hydromorphological attributes of ecological condition. Also defined as a parameter or value derived from parameters of a condition or process. An indicator is designed to facilitate the reliable tracking of progress towards an agreed target.

Intercalibration: A comparison and harmonization of the national interpretations of good ecological status under the European Water Framework Directive (EU WFD).

Invasive (introduced) species: Non-native species brought into an area intentionally or accidentally by humans and which develop negative impacts on endogenous species.

Lacustrine: Means “lake” or “relating to lake”.

Land cover: The physical coverage of land (e.g. initial vegetation, but also land use types are indicated), usually expressed in terms of vegetation cover or lack of it. Influenced by but not synonymous with land use.

Land use: The human utilization of a piece of land for a certain purpose (such as irrigated agriculture or recreation). Influenced by but not synonymous with land cover.

Least disturbed condition (LDC): is found in conjunction with the best available physical, chemical, and biological habitat conditions given today’s state of the landscape (Stoddard *et al.,* 2006).

Lentic: Related to slow-moving water, such as in lakes and bogs.

Lotic: Related to fast-moving water, such as in most streams and rivers.

Macroinvertebrates: Macroscopic animals without backbones (“invertebrates”) that are large enough to be seen with the naked eye (“macro”, e.g. > 0.5 mm).

Mesocosm: An experimental system designed to simulate natural conditions and use naturally occurring organisms and artificial structures.

Metric: A numerical measure known to monotonously respond to the degree of human-induced impact. With respect to ecological assessment, a metric is a communities’ attribute that is suited to measuring the degradation (e.g. number of taxa, proportion of current preferences, certain sensitive species, a saprobic index, etc.).

Monitoring: Is a time series of measurements of physical, chemical, biological and hydromorphological variables, designed to answer questions about environmental change (Lovett *et al.* 2007).

Multiple stressors: Are co-occuring stressors which may interact synergistically (i.e. in a self-energizing way), antagonistically (i.e. attenuating each other) or just neutrally with no direct relationships. For example, intensive land use for agriculture may cause water pollution and abstraction, with stronger pollution effects (i.e. less dilution) under reduced discharge.

Naturalness: It implies the absence of significant human disturbance or alterations.

Palustrine: Relating to a system of inland, non-tidal wetlands characterized by the presence of trees, shrubs and emergent vegetation (vegetation that is rooted below water, but grows above the surface). Palustrine wetlands range from permanently saturated or flooded land (as in marshes, swamps and lake shores) to land that is wet only seasonally.

Parameter: Selected chemical, physical, biological or other attributes used to characterize the qualitative and/ or quantitative status of water and aquatic ecosystems (the term "indicator" is also used in this respect). Also defined as a numerical or other measurable factor forming a set that defines a system or sets the conditions for its operation.

Pelagic: Living, swimming and feeding in the water column, as opposed to living associated with a sea or lake bottom.

Perennial: A perennial stream or perennial river is a stream or river (channel) that has continuous flow in parts of its stream bed all year round during years of normal rainfall. "Perennial" streams are contrasted with "intermittent" streams which normally cease flowing for weeks or months each year and with "ephemeral" channels that flow only for hours or days following rainfall. During unusually dry years, a normally perennial stream may cease flowing, becoming intermittent for days, weeks or months depending on the severity of the drought. The boundaries between perennial, intermittent and ephemeral channels are indefinite, and subject to a variety of identification methods adopted by local governments, academics, and others with a need to classify stream-flow permanence.

pH: In chemistry, pH is the negative of the base-10 log of the activity of the hydrogen ion in an aqueous solution.

Physico-chemical: Refers to a physical characteristic of water which derives from a single or combination of chemical constituent(s) of water (e.g. conductivity, redox potential).

Point pollutant sources: Discharges of pollutants from known discrete sources, e.g. an effluent discharge from an industry. The volume and quality of the discharge can normally be measured.

Policy maker: A person with power to influence or determine policies and practices at an international, national, regional or local level (MEA*,* 2003).

Precautionary principle: The management concept stating that in cases "where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation," as defined in the Rio Declaration 1992 (MEA*,* 2003).

Pressure: Pressures are the consequences of human activities seeking to satisfy various dimensions of human well-being. Through feedback loops, these pressures influence freshwater ecosystems. When these feedback loops are overlooked, pressures can reach levels beyond the natural resilience of the respective ecosystems. Their functions are impacted and they start to deviate from their 'healthy state'. Thus the pressures become sources of stress. See related term - **Stressor**.

Provisioning services: The products obtained from ecosystems, including, for example, genetic resources, food and fiber, and fresh water (MEA*,* 2003).

Raw water: Source water or water withdrawn from freshwater ecosystems in its untreated state).

Redox potential: An expression of the oxidising or reducing power of a solution relative to a reference potential. This potential is dependent on the nature of the substances dissolved in the water as well as on the proportion of their oxidized and reduced components).

Reference (condition): Natural or near-natural status, characterized by least impairment due to human activities, such as agriculture, settlement, organic pollution, eutrophication, water abstraction, etc. For any water body type or river type, reference conditions or "high ecological status" is a state in the present or in the past where there are no, or only very minor, changes to the values of the hydromorphological, physico-chemical and biological quality elements which would be found in the absence of anthropogenic disturbance.

Reference distribution: The reference distribution describes the variations of a measurement or value in a healthy aquatic system.

Regulating services: The benefits obtained from the regulation of ecosystem processes, including, for example, the regulation of climate, water, and some human diseases (MEA, 2003).

Resilience: An ecosystem´s ability to recover and retain its structure and function following a transient and exogenous impairment. If a stressor or disturbance does alter the ecosystem, then it should be able to bounce back quickly to resume its former ability to yield a service or utility rather than transform into a qualitatively different state that is controlled by a different set of ecosystem processes.

Response: The policy, measures and actions to improve the state of a water body (e.g. regulation and restriction of water abstraction, limitation of point source discharge, development of best practice measures, restoration schemes, water quality management, agricultural and fishery policies (EC, 2003).

Restoration measure: Activity to improve the status of degraded waters either by wastewater treatment or by structural improvement measures.

Salinity: A measure of the salt content of soil or water.

Sediment: The silt, sand, rocks, wood and other solid material that gets washed out from some places and deposited in others.

Species: Any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife that interbreeds when mature. A group of living organisms consisting of similar individuals capable of exchanging genes or interbreeding.

Species (taxa) richness: The number of species (taxa) present in a defined area or sampling unit.

Species diversity: An ecological concept that incorporates both the number of species in a particular sampling area and the evenness with which individuals are distributed among the various species. See also related term - **Biodiversity**.

Standard: An officially established and enforced set of indicators and indices together with their respective threshold values used to classify the aquatic ecosystems or natural water bodies. Standards may also include a description of how the respective indicators should be monitored, reported and evaluated. Standards are usually issued by competent governmental or intergovernmental agencies. Standards may also specify their area of validity and a potential timetable for achieving certain values of different parameters/ indices.

State: The condition of a water body resulting from both natural and anthropogenic factors (e.g., hydrological, morphological, chemical and biological conditions).

Stressors: Are concrete and negative manifestations of pressures or combinations thereof on inland waters such as construction of water infrastructure, modification of aquatic habitats, biological pollution, such as the emergence of invasive species, overexploitation of aquatic resources, chemical and thermal pollution, and flow alteration. In extreme cases even climate change can become a direct stressor.

Stressor gradient: A gradient describing the different levels of impact caused by a pressure (stressor).

Supporting services: Necessary ecosystem functions and processes for the production of all other ecosystem services. Some examples include biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling, and provisioning of habitat (MEA*,* 2003).

Surface water: All moving and standing water naturally open to the atmosphere.

Taxa (sing.: Taxon): A group of any size used in the classification of things, particularly plants and animals.

Threshold and threshold value: A certain concentration which delineates different categories of water quality / ecosystem health. Specific levels of water quality which, if transgressed, are expected to render a body of water unsuitable for its designated use.

Toxic substance or toxicant: Natural or synthetic chemical substance that may cause adverse effects on living organisms, even when present at low concentrations.

Trigger value: Trigger values are concentrations that, if exceeded, would indicate potential environmental problems, and so "trigger" a management response.

Typology: The study and interpretation of types. A typology provides the framework to group water bodies into appropriate types (streams, rivers, lakes, etc.). A stream typology covers all stream types encountered in a pre-defined region (usually an entire country or river basin) and provides supplementary data to describe the abiotic and biotic characteristics of the stream types . A particular type of rigorous classification in which a field of data is divided up into categories that are all defined according to the same set of criteria and that are mutually exclusive. See related term – **Classification**.

Uncertainty: An expression of the degree to which a future condition (e.g. of an ecosystem) is unknown. Uncertainty can result from lack of information or from disagreement about what is known or even knowable (epistemological uncertainty). It may have many types of sources: From quantifiable errors in the data to ambiguously defined terminology or uncertain projections of human behaviour. Uncertainty is also a consequence of the stochastic nature of processes and phenomena (aleatory uncertainty).

Utilitarian: An approach that focuses on the satisfaction of human preferences. In some cases, this is taken further and made the basis of a moral viewpoint.

Water quality: The term water quality is used to describe the physical, chemical, biological, hydromorphological and aesthetic properties of water that determine its fitness for a variety of uses and for the protection of the health and integrity of aquatic ecosystems. Many of these properties are controlled or influenced by constituents that are either dissolved or suspended in water.

Wadeable: Water courses from where samples can be taken by wading of the observer across the stream bed.

Water body: Distinct and significant volume of water. For example, for surface water: A lake, a distinct part thereof, a reservoir, a river or part of a river, a stream or part of a stream. For groundwater: A distinct volume of water within one or more aquifers.

Watershed: A planning term that refers to the area from which surface water drains into a common lake or river system or directly into the ocean; also referred to as a drainage basin or catchment basin.

Well-being: A context- and situation-dependent state, comprising basic material for a good life, freedom and choice, health, good social relations and security (MEA*,* 2003).

List of Abbreviations

AEV: Acute Effect Values
ANA: National Water Agency of Brazil
ANZECC: Australian and New Zealand Environment and Conservation Council
AUSRIVAS: Australian River Assessment System
BAC: Best Attainable Condition(s)
BAP: Biologically Available Phosphorus
BCG: Biological Condition Gradient
BLM: Biotic Ligand Model
BNDES: Brazilian Development Bank
BOD: Biological Oxygen Demand
BQE: Biological Quality Elements
BS: British Standards
CADDIS: Causal Analysis/Diagnoses Decision Information System
CCC: Criteria Continuous Concentration
CCME: Canadian Council of Ministers of the Environment
CEN: European Committee for Standardization
CETESB: Sao Paulo Environmental Agency
CEV: Chronic Effect Values
Chl-*a*: Chlorophyll-*a*
CMC: Criteria Maximum Concentration
COD: Chemical Oxygen Demand
CV: Coefficient of Variation
CWA: Clean Water Act (United States)
DO: Dissolved Oxygen
DOC: Dissolved Organic Carbon
DPSIR: Drivers, Pressures, States, Impacts, Responses Model
DPSSIR: Drivers, Pressures, Stressors, States, Impacts, Responses Model
DQO: Data Quality Objectives
DRPC: Danube River Protection Convention
DWAF: Department of Water Affairs and Forestry (South Africa)
EC: European Commission
EEA: European Environmental Agency
EFR: Environmental Flow Requirements
ELOHA: Ecological Limits of Hydrologic Alteration
ELV: Emission Limit Values
EMAP Environmental Monitoring and Assessment Program
EN: European Norm
EQR: Ecological Quality Ratio
EQS: Environmental Quality Standards
ES: Ecosystem Services
ESA: Ecosystem Service Approach
ESAWADI: Ecosystem Services Approach for the Water Framework Directive Implementation
ESU: Evolutionary Significant Union
ET Early Trigger
EU: European Union
EU WFD: European Water Framework Directive (Directive 2000/60/EC of the European Parliament)
FEPA: Federal Environmental Protection Agency (Nigeria)
GBIF: Global Biodiversity Information Facility
GDP: Gross Domestic Product
GEF: Global Environment Facility
GEO BON: Group on Earth Observations' Biodiversity Observation Network
GIG: Geographical Intercalibration Groups
GIS: Geographic Information System
GIWP: General Institute of Water Resources and Hydropower Planning and Design (China)
GMS: Governance and Management System
GPA: Global Programme of Action for the Protection of the Marine Environment from Land based Activities of UNEP
GWSP: Global Water Systems Project
HC05: 5th Percentile Hazardous Concentration
HDG: Human Disturbance Gradient
HEVAE: High Ecological Value Aquatic Ecosystems
IBI: Index of Biological Integrity
ICPDR: International Commission for the Protection of the Danube River
IECA: Integrated Ecological Condition Assessment
IPBES: Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services
ISO: International Organization for Standardization
IUA: Integrated Unit of Analysis
IUCN: International Union for Conservation of Nature
IWQGES: International Water Quality Guidelines for Ecosystems

IWRM:	Integrated Water Resources Management
KBA:	Key Biodiversity Areas
KNP:	Kruger National Park
LDC:	Least Disturbed Condition
LIPI	Indonesian Institute of Sciences
MAB:	Man and the Biosphere Program of UNESCO
MACs:	Maximum Allowable Concentrations
MDBA:	Murray-Darling Basin Authority
MDC:	Minimal Disturbed Condition
MDG:	Millennium Development Goals
MoE:	Ministry of Environment (and Forestry)
MoNRE:	Ministry of Natural Resource and Environment
MQI	Morphological Quality Index
MQO:	Measurement Quality Objectives
MRB:	Mekong River Basin
MRC:	Mekong River Commission
NEMA:	National Environment Management Authority (Kenya)
NGO:	Non-Governmental Organizations
NL:	Numerical Limit
NOEC:	No Observable Effect Concentration
NWA:	National Water Act
NWQMS:	National Water Quality Management Strategy
OECD:	Organisation for Economic Co-operation and Development
OP:	Operative Paragraph
PA DEP:	Pennsylvania Department of Environmental Protection
PJV:	Porgera Joint Venture
RBA:	Rapid Biological Assessment
RBMP:	River Basin Management Plan
RBO:	River Basin Organization
RBP:	Rapid Bioassessment Protocoll
RC:	Reference Condition
RDM:	Resource-Directed Measures
REACH:	Registration, Evaluation, Authorisation and Restriction of Chemicals
RHP:	South African River Health Programme
RIVPACS:	River Invertebrate Prediction and Classification System
RQO:	Resource Quality Objectives
RU:	Resource Unit
RV:	Reference Value
SD:	Standard Deviation
SDC:	Source-Directed Control Measure
SDG:	Sustainable Development Goals
SDI:	Spatial Data Infrastructure
SI:	Stressor identification
SIGNAL:	Stream Invertebrate Grade Number – Average Level.
SPM:	Suspended Particulate Matter
SPMR:	Sao Paulo Metropolitan Region
SSD:	Species Sensitivity Distributions
SUMHA:	Sustainable Management of Hydrological Alterations
TAN:	Total Ammonia Nitrogen
TDS:	Total Dissolved Solids
TMDL:	Total Maximum Daily Load
TN:	Total Nitrogen
TNMN:	Trans National Monitoring Network
TP:	Total Phosphorus
TPA:	Thematic Priority Area
TSS:	Total Suspended Solids
TV	Trigger Value
TWQR:	Target Water Quality Range
UAV:	Unmanned Aerial Vehicles
UFZ:	Environmental Research Centre of the Helmholtz Society, Leipzig, Germany
UK:	United Kingdom
UN:	United Nations
UNEA:	United Nations Environment Assembly (governing organ of UNEP)
UNECE:	United Nations Economic Commission for Europe
UNEP:	United Nations Environment Programme
UNESCO:	United Nations Educational, Scientific and Cultural Organization
UNGA	The United Nations General Assembly
UNSC:	United Nations Statistical Commission
UNU-EHS:	United Nations University, Institute for Environment and Human Security
UN-Water:	United Nations inter-agency mechanism on all freshwater related issues, including sanitation
UNWC:	UN Watercourses Convention
US:	United States of America
US EPA:	United States Environmental Protection Agency
WHO:	World Health Organization
WMA:	Water Management Area
WRC:	Water Resource Classification
WQAA:	Water Quality Assessment Authority (of India)
WQB:	Water Quality Benchmarks
WQC:	Water Quality Criteria
WQG:	Water Quality Guideline
WQI:	Water Quality Index
WQS:	Water Quality Standard
WRMA:	Water Resources Management Authority of Kenya
WSA:	Wadeable Streams Assessment
WSSCC:	Water Supply and Sanitation Collaborative Council
WWF:	World Wide Fund for Nature
WWT:	Waste Water Treatment
WWTPs:	Waste Water Treatment Plants

1 Introduction

Context and relevance of developing water quality guidelines for ecosystems

Biodiversity is widely regarded as an implicit measure to indicate the integrity and healthy functioning of ecosystems. Freshwater biodiversity is in serious decline (Strayer and Dudgeon, 2010; Dudgeon et al., 2006). While aquatic ecosystems contain a disproportionately high biodiversity, the Millennium Ecosystem Assessment (MEA, 2005) revealed that biodiversity decline in freshwater systems is occurring twice as fast compared to other ecosystem types. Their ability to provide Ecosystem Services (ES) is similarly in decline. This perspective is even more dramatic when put into a global context. The present rate of annual extinction of all kinds of species (freshwater, terrestrial and marine) can serve as an indicator of biodiversity loss. This is at present more than tenfold higher than the estimated, scientific consensus-based respective planetary boundary (Rockström et al., 2009). While both the concept and the suggested planetary boundaries are subject of scientific debate (Lewis, 2012; Nordhaus et al., 2012), the dramatic state of freshwater bodies, as arguably the most vulnerable ecosystems globally, is indisputably evident. In addition, ES have been considered as “services for free” for too long. Overstressing the resilience of freshwater ecosystems has led to massive deteriorations with major consequences for human health, livelihoods and biodiversity.

Deteriorating water quality also affects the availability of water for people, as a significant part of the resource can no longer be considered for higher value uses. Obviously, “business as usual” is not an option for sustainable water resources management. The situation has become alarming in many parts of the world, leading to an increased demand for immediate action and solutions to tackle the problems.
There is growing awareness that the water requirements to sustain ecosystem health and biodiversity in open surface waters and wetlands can be well aligned with human needs and the delivery of a range of ecosystem goods and services to society (Postel and Richter, 2003; Bernhardt et al., 2006). Indeed, the coexistence between the needs of ecosystems and those of human society is central to the concept of environmental flows, which describe “the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems” (Brisbane Declaration 2007; Matthews et al., 2014). It has also prompted increased attention to the need for far greater alignment in the future between the processes of environmental flow assessment and the setting of water quality standards for freshwater ecosystems.

In developed countries, where the problems had become very acute and where financial resources are available, considerable investments in catchment protection and restoration, in water quality improvements (Zhang and Wen 2008), and in environmental flow provisions (Tharme, 2003; Poff et al., 2010) have been made to reduce the risks to freshwater ecosystems and to protect drinking water sources. However, for countries which may still have the choice, a more precautionary and cost-effective approach would be required to identify and tackle the causes of degradation at their source (Vörösmarty et al., 2010). As prevention is more effective and usually considerably less expensive than the cure, adopting pollution-prone development models is counterproductive.

Investments in the protection of water resources and, where necessary, restoration to improve the condition of degraded water quality and ecosystems must be undertaken as part of an adaptive process (Walters, 1986). This process is underpinned by an agreed set of values and objectives, is informed by new science, guides management actions, and is responsive to the changing perceptions and values of stakeholders (Bunn et al., 2010). It also requires extensive consultation and engagement with interest groups to derive a shared understanding of the range of important environmental assets and values that need to be considered (Meyer, 1997; Walters, 1997).

Monitoring and evaluation is an essential element of the adaptive management process to ensure that management interventions are successful and environmental objectives are met (Lovett et al., 2007; Pahl-Wostl et al., 2007; Bunn et al., 2010). Accordingly, significant efforts have been made to develop cost-effective tools and methods to monitor and evaluate the success, or otherwise, of management investments (Friberg et al., 2011). These efforts include setting up water quality guidelines for freshwater ecosystems.

Status quo of water quality guidelines for freshwater ecosystems

Water quality, biological and hydromorphological conditions of freshwater bodies not only characterize the status of freshwater ecosystems, but reflect the prevailing situation in neighbouring terrestrial ecosystems as well. As ultimate sinks in the landscape (through surface runoff and exchanges with groundwater bodies), freshwater ecosystems are excellent proxies to characterize the ecological health of upstream areas and entire river basins.

This potential is, however, not yet fully utilized. While international and national Water Quality Guidelines (WQGs) (with a utilitarian focus) already exist for drinking water[3], recreational use[4], irrigation[5], livestock[6] and water reuse[7] (among others), comparable WQGs for ecosystems with a focus on the ecosystem health of inland waters are rare. Besides the “utilitarian” water quality standards, similar regulatory mechanisms are needed for freshwater ecosystems. These would provide an appropriate framework and basis for the remediation and monitoring of water bodies, ultimately ensuring freshwater ecosystem health and function, including the provision of ES. These will be important to assess whether countries are meeting the SDG targets 6.3, 6.6 and 15.1 set for the protection of freshwater ecosystems and the reduction of water pollution. Water quality standards for ecosystems would facilitate the integration of an ecosystem-based management approach (one which considers ecosystems as legitimate water users, if not the resource base itself) in water resources management.

As a primary prerequisite for the sustainable restoration/rehabilitation actions and environmental safeguards, water quality standards for ecosystems are necessary. The diversity and complexity of freshwater ecosystems presents challenges to the development of such international water quality standards. The natural interrelationships between water quality and water quantity to ensure ecosystem health are seldom made explicit. Thus, this scientific background volume on how to establish standards for the health of freshwater ecosystems include, in addition to the general principles, recommendations from quality and quantity perspectives including physical, chemical, biological and hydromorphological features. WQGs, predominantly based on physical and chemical parameters, to protect freshwater biota are well developed in some regions (e.g. Australia and New Zealand; US; Europe; South Africa). However, it is generally well accepted that these measures alone cannot ensure the protection of biodiversity and the maintenance of

[3] http://www.who.int/water_sanitation_health/dwq/guidelines/en/
[4] http://www.who.int/water_sanitation_health/bathing/srwe1/en/
[5] http://www.oecd.org/tad/sustainable-agriculture/waterqualityandagriculturemeetingthepolicychallenge.htm
[6] http://www.fao.org/DOCReP/003/T0234e/T0234E08.htm;
http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX%3A31991L0676
[7] http://ec.europa.eu/environment/water/reuse.htm; http://www.unep.or.jp/Ietc/Publications/Water_Sanitation/wastewater_reuse/

essential ecological processes. Consequently, the use of biotic indicators has become a core component of freshwater ecosystem health assessment. To date, biotic indicators have been especially integrated in freshwater assessment in the developed world (e.g. Wright, 1995; Schofield and Davies, 1996; Rosenberg et al., 1997; Barbour et al., 1999; Hering et al., 2010), but also increasingly in emerging economies and developing countries for instance in South Africa (DWAF, 1996a).

For this reason, the current guidelines go beyond the consideration of 'traditional' physico-chemical water quality indicators and include the consideration of biological and hydromorphological indicators to provide a more comprehensive assessment of freshwater ecosystem health. It is worth noting that the inclusion of biotic indicators also provides a sensitive measure of the risks to drinking water quality and has the added advantage that freshwater organisms are continuously exposed to stressors and therefore provide real-time responses to these stressors, indicate accumulation of impacts and integrate pressures and stressors over longer time-spans – unlike many spot tests for chemical parameters.

Objectives and scope

Scope and intergovernmental mandate for developing water quality guidelines for ecosystems

The objectives and scope of this work are influenced by the UN Environment Governing Council Decision 27/3 "to develop international water quality guidelines for ecosystems that may be voluntarily used to support the development of national standards, policies and frameworks taking into account existing information while integrating, as appropriate, all relevant aspects of water management". Recognising complexity, uniqueness and variation of freshwater ecosystems around the world, and the challenges associated with determining water quality guidelines for ecosystems, a framework has been developed to guide countries through the key processes to be able to determine water quality guidelines for ecosystems in their own national contexts. This framework is known as the Framework for Freshwater Ecosystem Management (see Preface and Acknowledgements sections for more information on the volumes in the Freshwater Ecosystem Management Series).

The Framework includes guidance on water quality guidelines for specific parameters, based on a review of existing water quality guidelines (see Chapter 3).

The Framework aims to assist policy makers at international and national levels and the respective governance and management authorities in the establishment of appropriate regulatory mechanisms enabling the sustainable management of their water resources. Strong focus on the challenges of the protection of water quality, and its relationship with water quantity, for freshwater ecosystems is of fundamental importance.

It is necessary to emphasize the direct and indirect provisioning of ES, in particular from freshwater ecosystems as they constitute the life support for the planet including its human population. Without ensuring the sustained health and natural functions of the respective ecosystems, these services cannot be relied upon. Given the alarming state and prevailing deteriorating tendencies of the quality of water in various ecosystems in many parts of the world, there is an urgent need for the Framework to play an effective role in attaining future water security in its broadest sense.

The Framework focusses on water quality conditions that sustain healthy freshwater ecosystems within the various forms of inland (surface) waters. It also draws on linkages between the provisioned ES and respective human uses. The Framework is intended for global relevance, with varying local/ regional water quality changes, although it is expected that countries that have not yet put in place regulatory activities for the protection of water quality and freshwater ecosystem health management will benefit most. Therefore, a strong focus is on assisting developing countries in improving the protection of their freshwater resources.

In the context of this volume, the term “Guidelines” refers to a document conceived and compiled as a roadmap including recommendations helping the respective government authorities to establish guidelines or (potentially binding) standards for water quality requirements for freshwater ecosystem health within their respective jurisdiction.

The Water Quality Guidelines within this Framework are intended to be science-based recommendations, hence providing, among others, a set of non-prescriptive physical, chemical, biological and hydromorphological characterizations of certain categories of freshwater ecosystems hosted by inland water bodies. The Framework subscribes to the precautionary principle to approach uncertainties in data and knowledge of ecosystem processes. As the Framework is of global relevance but has to account for varying local and regional challenges, it is necessarily broad, having a coarse resolution. It advocates an iterative, adaptive management[8] approach to revisit the perceived challenges, but also to modify policies, objectives and the approach cyclically towards the achievement of the intended goal.

Corresponding with this philosophy, the Framework is conceived as a “living document” that will need further development, refinement and validation on regional, national and sub-national scales, considering local ecotypes, hydrological conditions, stakeholder aspirations and contexts. The phases of regional consultations after 2015 and the potential revision(s) of the Framework after its adoption by the UN Environment Assembly in 20117 are foreseen to account primarily for these needs.
The Framework provides the most relevant information required to develop WQGs for freshwater ecosystems including approaches to identify indicators and set target and threshold values. These steps should be synchronized with the agreed indicators for monitoring the achievement of the SDGs, in particular SDG targets 6.3, 6.6 and 15.1 (UNGA, 2015). Thus, the Framework provides a sound base on which to build more region-specific and national, strategically targeted programmes.

[8] Adaptive management refers to a systematic process for continually improving management policies and practices by learning from the outcomes of previously employed policies and practices.

The Framework: a guide to act

All nations and other stakeholders who are legitimate custodians and users of freshwater resources within their respective jurisdictions must agree on a set of standards and objectives for these waters at the respective - usually national - level. Whether ecosystem functions and/or ecosystem services provided by a certain river or lake remain intact or are being sustainably restored is ultimately a political (preferably multi-stakeholder) decision. Nevertheless, the scientific community and intergovernmental agencies can provide information and guidance on this process. Countries can then promulgate their respective (binding) water quality standards based on scientifically-sound and regionally-relevant recommendations. However, the Water Quality Guidelines in this volume are not prescriptive and, while suitable indicators and threshold values are recommended in the document, countries need to adapt and choose their own indicators and thresholds to address regional water quality objectives and to take into account regional conditions. These set thresholds and targets should be realistically achievable.

While considering and drawing on available relevant guidelines, standards and scientific literature, this volume does not advocate the simple transfer of available knowledge to unchartered geographical locations. The objective is therefore rather to define regionally-relevant principles and advise how to establish binding standards that will guide and support the formulation of locally-relevant policies, the protection and rehabilitation towards improving freshwater ecosystem health. This approach is to be synchronized with the relevant targets and indicator monitoring and reporting of the SDGs.

Thus, the set of indicators and thresholds presented in the Framework will serve as well-founded scientific advice. As such, they are not a substitute for standards to be established and enforced by sovereign state authorities or by intergovernmental bodies (such as the EU) or through international conventions to be observed within their respective jurisdictions, according to implementation time schedules formulated by the same authorities and/or their political organs.
Thresholds, their definitions and numerical values are frequently the subject of scientific and political debate. Owing to their implications for economic activities and consequences for potential remedial actions, they may also have to be part of a broader societal debate. Hence, the Framework serves as the basis for the process of setting national standards both as a guide for administrative and as a roadmap to reach societal consensus.

Different spatial scales and classification schemes may need to be utilized to define relevant water quality thresholds for various freshwater ecosystems. However, due to limited baseline data about the health and functioning of freshwater ecosystems and due to the lack of, or sparse, monitoring networks in many regions, it is expected that some recommendations of Water Quality Guidelines may have to be based on best professional estimates. In this context, this volume identifies the need for more comprehensive and targeted monitoring programmes and further research, as outlined in »Chapter 6.

Structure of Volume 4

This Scientific Background volume contains methodological and technical details related to different phases of water quality management, guideline development, implementation and enforcement. It is structured as follows:

Chapter 2 presents the conceptual framework and philosophical and scientific basis for developing Water Quality Guidelines for ecosystems. This includes the most pertinent principles, definitions and state-of–the-art methods used to set freshwater ecosystem health objectives, indicators and classifications. It addresses pressures, stressors and trade-offs emanating from simultaneous societal use and sustainable ecosystem functions of various water bodies as well as the planning and practical implementation of monitoring. Problems which may be encountered in practical applications such as uncertainty and data scarcity are also discussed. Establishing Reference Conditions (RCs) for freshwater ecosystems as well as achievable target conditions are outlined in both their policy-governance and scientific-technical dimensions. Reporting and communicating results to the public and policy makers are addressed as well as the legal and governance framework, capacity (institutional and professional personnel) constraints and the need for a phased, adaptive approach. Chapter 2 is meant as background reading first and foremost for professionals, scientists and persons engaged in water quality and ecosystem management. Chapter 2 also refers to ongoing research and conceptual development. While it reflects the state of the art, it is also acknowledged that some of the presented methodologies need more data, more elaborate monitoring and analytical capacities than feasible - as of now - in many countries. In this context, Chapter 2 outlines the perspective while the recommended approach to establish WQGs for freshwater ecosystems is presented in detail in Chapter 4.

Chapter 3 takes stock of existing knowledge by providing a structured overview of 15 selected water quality standards and guidelines for freshwater ecosystems including their guiding principles, legal status, governance structure, typology, classifications, reference and target setting from different regions of the world. Indicators and threshold values for different quality classes as used by those guidelines are described. These guidelines may serve as examples as well as lessons learnt for policy makers. Three advanced guidelines and standard systems for freshwater ecosystems, the EU Water Framework Directive (EU WFD), the joint Australian-New Zealand guidelines and that of the United States Environmental Protection Agency (US EPA), are discussed and presented in greater detail.

Chapter 4 outlines the suggested management framework for developing and applying water quality guidelines and describes how to proceed in setting up standards and guidelines for the water quality requirements of freshwater ecosystems. As far as the science-based recommendations for practical implementation are concerned, Chapter 4 is the core part of this volume. A “4 phases 9 steps” approach is developed and outlined[9]. Objective setting, classification of freshwater ecosystems, climate, regional specificities as well as pressures and stressors, human water use and other relevant water-related activities are among the starting points. They are followed by recommendations on how to select indicators and set numerical thresholds to distinguish ecosystem health classes and how to identify particular conservation areas. Defining RCs and/or approximating the Best Attainable Conditions (BAC) are addressed as both policy making and technical questions. Chapter 4 elaborates the phased and adaptive development concept of establishing WQGs for freshwater ecosystems. Essential features of this development such as cost-

benefit considerations, institutional and manpower capacity issues, stakeholder involvement, including citizen science approaches, are among the topics covered. This is linked to the “monitoring ladder” concept of Integrated Monitoring of Water and Sanitation[10] and the Joint

[9] This 4 Phase, 9 Step Framework is the precursor of the 4 Phase, 12 Step Framework later adopted, as described in the preface section of this volume.
[10] Integrated Monitoring of Water and Sanitation related Sustainable Development Goals Targets. Global Enhanced Monitoring Initiative (GEMI) is an inter-agency initiative composed of the United Nations Environment Programme (UNEP), the United Nations

Monitoring Programme for Water Supply and Sanitation[11].

Chapter 5 provides a number of examples from different continents and contexts which illustrate and validate the feasibility of the proposed “4 phases 9 steps” approach for the development of guidelines, but also as a general framework for the management and restoration of freshwater ecosystems.

Chapter 6 presents an outline of issues and recommendations for further follow-up development of the Framework.

This volume contains a unified list of references and 6 supporting annexes with methodological details and a comparative analysis of benchmarks and threshold values for key indicators characterizing and demarcating freshwater ecosystem classes in existing regional and/or national standards and guidelines. A Glossary and List of Abbreviations are also provided. Illustrative boxes are inserted into the text, alongside numerous tables and figures.

Human Settlements Programme, the United Nations Children’s Fund (UNICEF), the Food and Agriculture Organization of the United Nations, the United Nations Educational, Scientific and Cultural Organization, the World Health Organization and the World Meteorological Organization, co-operating under the umbrella of UN-Water.

[11] WHO/UNICEF Joint Monitoring Programme for Water Supply and Sanitation

2 Principles and concepts: the philosophy and scientific basis of the Framework for Freshwater Ecosystem Management

Important properties of inland surface waters

Properties of inland surface waters

Inland surface waters have a number of distinguishing characteristics in that they:

- are linear or mosaic features embedded into the terrestrial matrix;
- are typically located at the topographically lowest point in the landscape, thereby collecting and conveying materials (water and dissolved and particulate matter) from within their entire catchment;
- may expand, contract and fragment, leading to rapid changes in volume and/or area;
- exhibit fast ecological succession processes; and
- are closely linked to and mutually dependent on adjacent terrestrial (surface and subsurface) and, in many cases, marine systems.

These unique properties make inland surface freshwaters among the most complex, dynamic and diverse ecosystems globally. This has major ramifications for management: The catchment (or basin) is the key unit for conservation and management. Connectivity within the freshwater ecosystem – longitudinally, laterally and vertically – is fundamental in understanding and managing inland surface waters. In river drainage networks, this influence of the land on freshwater systems is strongest in fringing or riparian lands (Alan, 2004; Peterson et al., 2011) or through hydrological connections with the floodplain (Tockner et al., 2008). Most freshwater ecosystems are disturbance-driven systems shaped by hydrological, morphological and biological events. For example, hydrological connectivity, the water-mediated transfer of energy, matter and organisms among and between the elements of the hydrological cycle, controls biodiversity and ecosystem processes and services on the catchment scale (Bunn and Arthington, 2002).

Global distribution of inland surface waters

Globally, a total of 304 million natural lakes cover a combined area of 4.2 million km2 (Downing, 2009). The global network of all rivers and streams, defined as lotic systems with an average annual discharge of more than 1 m3/s, is 7.56 million km long (Lehner et al., 2011) and covers an area of about 508,000 km2 (Lehner et al., 2011). Rivers that are temporary and seasonally or intermittently cease to flow and dry have been described as being more common and thus more representative of the world’s river systems than those with perennial flows (Williams, 1988). These temporary waters are mostly neglected when assessing rivers and streams, even though in many parts of the world they are a common type of surface water (Acuna et al., 2014; Steward et al., 2012). Worldwide, there are about 2.8 million reservoirs larger than 0.01 ha (Lehner et al., 2011). The total storage volume of all reservoirs amounts to

8,000 km3 and their combined area covers 306,000 km2 (excluding regulated natural lakes), equivalent to an increase of Earth's naturally occurring terrestrial water surface by 7.3% (Downing et al., 2006). In addition, continental wetlands cover between 12 and 15 million km2 corresponding to 3% of the Earth's surface (Downing, 2009). This also implies that 8 – 10% of the landmass of the globe is covered by wetlands.

Inland water bodies as centres of biodiversity

Inland surface waters contain disproportionately more species per unit area than marine and terrestrial ecosystems. Although inland waters − excluding wetlands − cover only about 1% of the Earth's surface, they provide habitat for more than 10% of known animals and about one-third of all vertebrate species (Balian et al., 2008). Our current knowledge of freshwater species diversity varies greatly between groups of organisms, and existing diversity is very much underestimated. Even among freshwater fish, almost a hundred new species have recently been described per year in South America alone (Abell et al., 2008).

An almost unexplored group of freshwater systems in terms of biodiversity are the groundwater-dominated subterranean water bodies underlying many regions (e.g. the underground rivers and cenotes of the Yucatan Peninsula of Mexico and karst system caves of Slovakia). An estimated 50,000 to 100,000 stygobiont species, i.e. species that finish their entire life cycle in the subterranean freshwater realm, occur globally (Culver and Holsinger, 1992). However, fewer than 10% of these species have been described up until now (Stoch and Galassi, 2010). Groundwater systems are characterized by a very high proportion of endemic and cryptic species, although there is a major lack of information on ecology and their functional importance.

Contemporary biodiversity distribution

Freshwater biodiversity is generally neglected or underrepresented in data collection efforts. The importance of broad biodiversity compilations, however, has been increasingly recognized, especially in light of the establishment of the Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (IPBES) and the Group on Earth Observations Biodiversity Observation Network(GEO BON). The Global Biodiversity Information Facility (GBIF; http://www.gbif.org) collates and centralizes biodiversity information through its participant nodes. BioFresh (http://www.freshwaterbiodiversity.eu), an EU–funded project, serves the same purpose for the freshwater realm (see case study below). Synthesis of species, genes and habitats distributions and hotspots of endangerment are critical for setting conservation priorities (Feeley and Silman, 2011). Such a synthesis requires continuous efforts to access and unite widely dispersed biodiversity data and to establish open data archiving as a standard scientific practice.

Key Freshwater Biodiversity Areas support conservation and management strategies

Holland et al. (2012) described a methodology for identifying priorities for freshwater protected areas via the development of freshwater Key Biodiversity Areas (KBAs). KBAs are defined as the presence of threatened and endemic species or ecologically unique assemblages of species, and are mapped using HydroBASINS (Lehner, 2012). HydroBASINS is a publicly available, digital hydrology resource for mapping connectivity within catchments, incorporating river basin boundaries, lakes and river networks . (Lehner 2012).
The application of these methods to various continents and regions (Allen et al., 2010, 2012; Darwall et al., 2011b; Molur et al., 2011) has identified a large number of potential KBAs (e.g. »Figure 2.1), which again may be compared to protected areas to identify gaps in both spatial coverage and management focus[12].

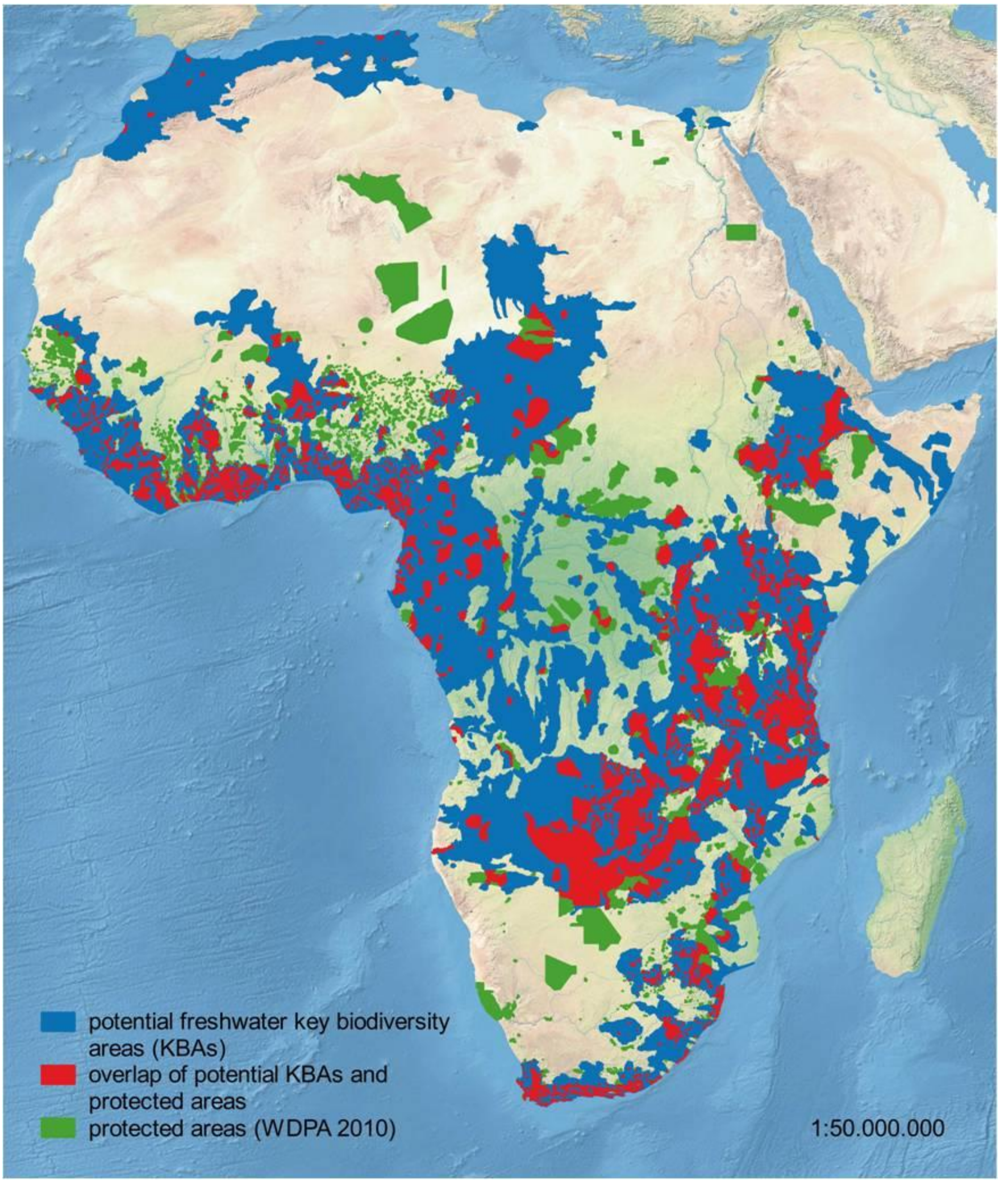
Long-term environmental and biodiversity trends

Long-term ecological studies are critical for providing key insights into environmental change, natural resource management and biodiversity conservation. However, there is a major gap in the availability of hydrological, physico-chemical and biological long-term data. Inland surface

[12] http://www.hydrosheds.org/page/hydrobasins

waters are hotspots of endangerment and there are clear signs that freshwater biodiversity is declining rapidly (Dudgeon et al., 2006; Darwall et al., 2009; Mittermeier et al., 2010). Population trend data indicates that, whereas terrestrial species show declines in the order of 25% since 1970, the equivalent value for freshwater species is 55% (Loh et al., 2005). This population trend data is based entirely on a selection of water-associated vertebrates and lack adequate representation from the more species-rich invertebrates (Cardoso et al., 2011;

Figure 2.1: Key Biodiversity Areas (KBAs) in Africa and the network of protected areas. Source: Darwall (2013); Holland et al. (2011).



Balian et al., 2008). While existing knowledge is inadequate, at least 10,000-20,000 freshwater species are estimated to have become extinct within the last century or are currently at risk globally (Strayer, 2006; Strayer and Dudgeon, 2010). About 30 % of all freshwater species that have been assessed by the International Union for Conservation of Nature (IUCN) are classified as threatened (i.e. ‘Critically Endangered’, ‘Endangered’ or ‘Vulnerable’ according to Red List criteria). Amphibians are the second most threatened group of organisms (after cycads) that have been assessed globally (IUCN, 2013); and, in intensively-developed regions, over one third of the species in other freshwater taxa are also threatened (e.g. Kottelat and Freyhof, 2007; Jelks et al., 2008; Cuttelod et al., 2011; Collen et al., 2014). Although knowledge of freshwater biodiversity is improving, information gaps remain especially for the species-rich tropical regions. Therefore, the overall threat extent might be even greater than currently estimated. »Box 2.1 provides examples of freshwater ecosystem databases and platforms allowing for access to all available databases describing the distribution, status and trends of global freshwater biodiversity.

WISE-WFD Database

A readily accessible example with respect to data compilation is WISE – the gateway to information on European water issues (http://water.europa.eu/). WISE is a partnership between the European Commission (Directorate-General for the Environment, Joint Research Centre and Eurostat) and the European Environment Agency (EEA). It comprises a wide range of data and information collected by EU institutions to serve several stakeholders across Europe. The on-going work of EEA is to transform the database from an information source to a knowledge source – what used to be mere statistics will be in-depth analyses of data in the near future.

The Freshwater Information Platform

(www.freshwaterplatform.eu) offers a forum for information exchange and open-access publishing of maps and data, and aims to stimulate cutting-edge research and collaborations in the field. The Platform provides a unique and comprehensive knowledge base for sustainable and evidence-based management of threatened freshwater ecosystems and the resources they provide.

Pressures such as discharge of polluted water, intense land use and climate change are increasingly threatening the health and diversity of European and global freshwater ecosystems. Over recent years, many European Union (EU) funded research projects have investigated the causes of these pressures and their effects on rivers, lakes and wetlands, and developed appropriate rehabilitation strategies. However, access to and use of the data generated by these projects is often difficult for water managers, policy makers, scientific communities and the general public. This is, because scientists have not yet fully adopted systematic data publishing practices. It is challenging to extract data embedded in the huge number of scientific papers and research project websites.

In order to make this detailed and wide-ranging knowledge of freshwater ecosystems accessible to all, four European research institutes in Austria, Belgium and Germany have joined forces to launch the Freshwater Information Platform (www.freshwaterplatform.eu), an interactive website integrating results and original data stemming from finished, ongoing, and future freshwater research projects.

The platform contains several complementary sections, either providing access to original data or summarizing research results in an easily digestible way. All sections are composed as ‘living documents’ that will be continuously improved and updated.

The “Freshwater Biodiversity Data Portal”, for example, provides access to data on the distribution of freshwater organisms (such as fish, insects and algae) both in Europe and worldwide; whilst the “Global Freshwater Biodiversity Atlas” provides a series of maps on freshwater biodiversity richness, threats to freshwaters (or ‘stressors’), and the effects of global change on freshwater ecosystems (http://atlas.freshwaterbiodiversity.eu/).

The “Freshwater Species Traits Database” integrates knowledge on the ecology of around 20,000 species inhabiting European freshwater ecosystems, including information about where species live, what they feed on or how tolerant they are to pollution (http://www.freshwaterecology.info/).

The “Freshwater Metadata” section provides an overview of hundreds of major data sources related to freshwater research and management, and offers the option to publish such data in the Freshwater Metadata Journal (http://data.freshwaterbiodiversity.eu/).

Box 2.1 Example of freshwater ecosystem databases and information platforms

Typology of inland surface waters

What is the need for the development of a typology?

Freshwater ecosystems contain a great variety of inland water bodies, most of them surface waters. Therefore this volume, while focusing on freshwater ecosystems, also relies on the well-established scientific terminology and category of inland surface waters.
Large-scale (e.g. national or river basin) water quality monitoring and management programmes may need to cover a broad range of freshwater ecosystems, from small headwater streams to large rivers, lakes and wetlands, and flowing through tropical, temperate or desert environments. It is important to recognize these natural differences in water body types when developing a monitoring programme because:

- different types of inland water bodies will not look and function the same way even when they are healthy;
- the types of indicators that might be appropriate in one type of water body may not be appropriate for another;
- the methods used to sample one type may not be applicable or relevant to another; and
- even when the same indicator can be used in different inland water body types, the threshold or target values are likely to differ (Karr, 1999; Bunn et al., 2010; Hering et al., 2010).

The scale of the typology required will be influenced by the spatial extent of the monitoring programme (in turn, monitoring should be designed such that it reflects different types of water bodies well). For example, a coarse scale typology would be sufficient on a national scale but finer-scale may be required in a specific region (e.g. Moog et al., 2004). Rivers (or wetlands) and catchments that are characterized by similar properties and functionscan be aggregated into one type. The selection of indicators and threshold values for indices to demarcate different quality states can then be determined for each class of water body (Karr, 1999; Bunn et al., 2010).

Different river types has been widely used to underpin river health assessment and management (e.g. Harris et al., 2000; Snelder and Biggs, 2002; Bunn et al., 2010; Kennard et al., 2010; Zhang et al., 2012). Typology is an important step in freshwater ecosystem health assessment, and its subsequent management, as it ensures comparisons among similar types of systems and forms the basis for setting threshold for determining what is considered good health and what is not.

For the proposed Framework, we have opted for a coarse-scale typology[13], recognizing that a more specific typology will need to be developed for specific purposes and individual areas, reflecting the regional heterogeneity of water bodies.

General typology

Inland waters are water bodies on or in the terrestrial portion of the Earth. The hydrological network within catchments forms a network of, often complicated, interconnected water body types. They comprise four broad categories of natural water bodies and an additional category for man-made freshwater ecosystems:

- running or lotic water ecosystems (streams, rivers and their fringing riverine wetlands),
- standing or lentic water ecosystems (lakes, reservoirs, ponds and their fringing lacustrine wetlands),
- (palustrine) wetland ecosystems,
- groundwater ecosystems and
- artificial or heavily modified water bodies (e.g. reservoirs, canals, constructed

[13] Coarse-scale typology is often a map-based classification of types e.g. based on ecoregions and some specific water body features; however, the strength of classification into types increases with the inclusion of local-scale variables. But since biological assemblages are affected by large-scale, i.e. regional, as well as local-scale environmental factors, ultimately, both scales should be taken into account.

wetlands, ditches, gravel pits).

Figure 2.2 A Proposed typologyof inland waters. (A) Overall structure and detail for lentic water bodies. *Dotted boxes are not considered in this volume.*

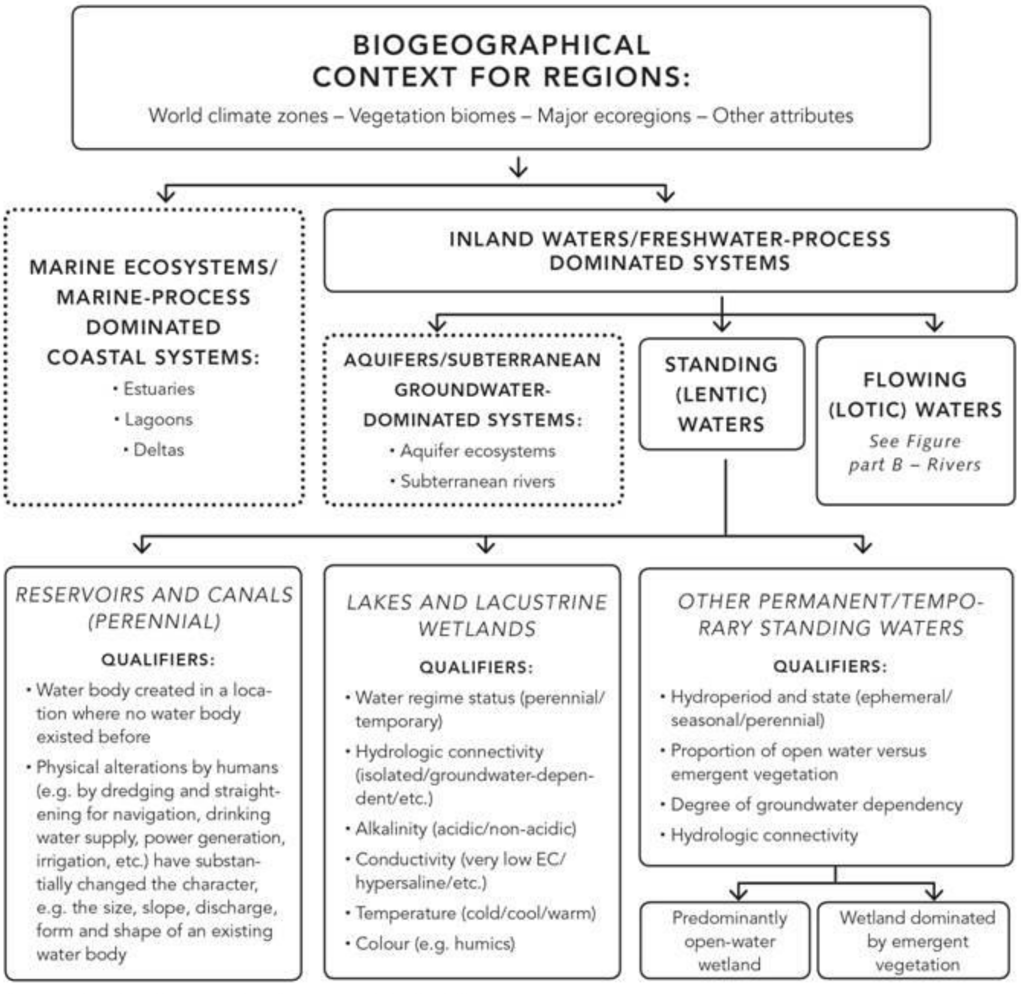


Figure part A – Classification of inland water types: Overall and Standing waters

This typology for inland water bodies provides a classification system that can be used within the Framework. Note that similar types could also be identified based on physical, chemical and related characteristics in the absence of an existing classification system. However, if a classification system for water body types already exists, the existing system should be fully considered and if possible used to ensure continuity with existing programmes and policies that are in place. If the existing system needs to be modified, all modifications should be thoroughly explained and documented. Artificially created water bodies or those whose physical structure has been heavily modified need to be included in these schemes so as to provide a comprehensive assessment and because they can form extensive components of a region´s waterscape.

Physical characteristics are fundamentally based on water residence time and regime (e.g. whether permanent or temporary, stable or fluctuating, running or standing) with further divisions determined by climatic factors, water body size and physico-chemical properties including hydro- and geomorphic features, and geochemical and thermal characteristics. Many chemical pollutants vary in their form and their potential to harm, depending on the overall environmental context. For example, bioaccumulation of heavy metals in food webs varies with pH and redox potential (Luoma, 1983).

Figure 2. B: Proposed typologyof inland waters. (B) lotic water bodies.

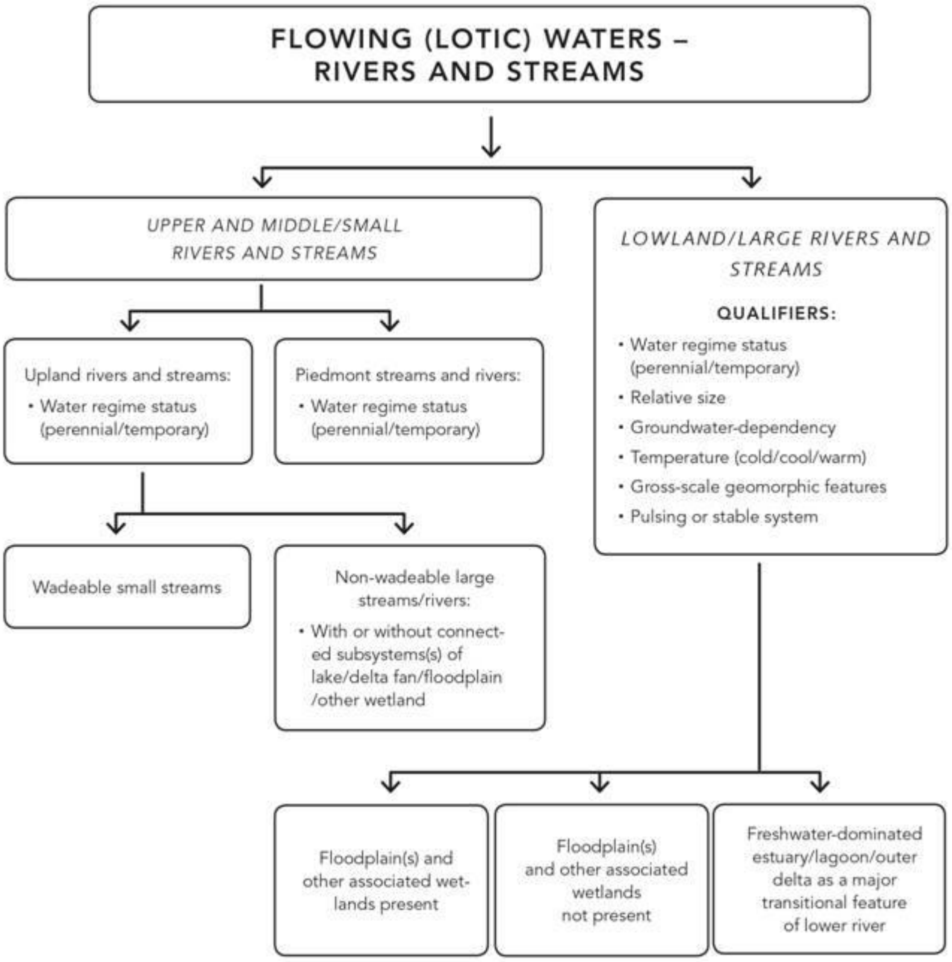


Figure part B – Rivers

Where a typology does not already exist to support the establishment of national guidelines, or to guide their development, a simple, globally generic and hierarchical inland water typology (»Figure 2.2) is proposed as a starting point. It is drawn largely from the basic typology in use in national guidelines (e.g. those of the EU, New Zealand and Australia) and informed by water quality schemes in many countries. The typology is designed to be an overarching framework that will cover most waters, yet remain compatible with national wetland inventories and more complex or elaborated typologies. It is structured to allow meaningful comparisons of water quality and ecological characteristics so that, in broad terms, “like is compared with like”. The typology provides a basic structure to guide assessment and monitoring.

General categories of water body types are, firstly, separated into those predominantly affected by marine influences (coastal lagoons and deltas) and by freshwater inflows. Non-marine dominated or inland waters may be flowing (lotic) or standing (lentic) waters, including wetlands, with subsequent divisions based on basic physical parameters (e.g. depth), location within the catchment, and fundamental physico-chemical properties. Owing to their widespread occurrence, general importance for water management and importance in many instances for biodiversity and ecosystem services (ES) (e.g. reservoir fisheries), the typology includes reservoirs, artificial channels and constructed wetlands. Groundwaters are included in the schematic diagram of »Figure 2.2 as a separate category. While groundwater systems are not explicitly addressed at present, the connectivity between surface waters and groundwaters is recognized. Groundwater contributions are, for instance, critical as the baseflow component of rivers and essential for various palustrine wetlands.

The different water body types reflect natural catchment geology, vegetation and climatic conditions, alongside primary hydrological and hydraulic characteristics of flow velocity, shape, turbulence, roughness of the stream bed and sediment regime. They include:

- Alkalinity. This is an indicator of water hardness (pH and conductivity) and high alkalinity reflects the presence of carbonate-rich geology.
- Colour. As a consequence of the drainage of organic rich soils, some systems have tea-stained colour and natural acidity (low pH) because of tannins and other organic acids.
- Suspended sediment. Some systems have naturally high concentrations of fine suspended sediment and water clarity is naturally low (e.g. white and black waters).
- Salinity. In arid regions, and particularly closed (endorheic) inland basins evaporation leads to high concentrations of dissolved salts – at times exceeding that of seawater.

Further regional development is possible to incorporate waters of special natural character or which reflect unique bio-geographical circumstances such as waters rich in endemic species typified by the world's network of ancient lakes or hypersaline lakes.

Geomorphic characterization

Rivers

A geomorphic classification of rivers is valuable for linking forms and processes. Stream order is among the most widely used descriptive classification of rivers, whereby the river network is divided into links between nodes. Stream order correlates with drainage area, slope, channel size as well as with chemical and biological conditions. Alluvial rivers can be classified according to channel types and patterns as straight, meandering and braided channels. The approach by Rosgen (1996) is commonly used for rapid, practical application. Eight stream types based on entrenchment, width-to-depth ratio and sinuosity are delineated. A nested hierarchical framework starting with catchment characteristics, landscape units, river types and geomorphic units is very useful in providing the biophysical basis to prioritize river management strategies.

Lakes

As for rivers, it is desirable to categorize different types of lakes to assist with monitoring and typology schemes, following the schematic outline in »Figure 2.2. The range of lake types is very extensive and variable, along continua of morphology, climate, connectivity and background geologically-derived water chemistry. A good summary of lake types is provided by Wetzel (2001). Ecological communities in lakes are further influenced across scales from local habitat structure to biogeography. Most lakes drain upstream catchments and discharge to downstream river or estuarine outlets. However, endorheic lakes (without an outlet), a feature of many semi-arid, land-locked regions, have inflows but no outflows. Such lakes are typically ephemeral (temporary) and of high salinity owing to gradual accumulation of salts. Lakes in northern and southern temperate climates are relatively recent features of the landscape, many formed from movement of ice during the last extensive periods of glaciation dating from 10-30,000 years ago. Many shallow lakes from this period have subsequently filled with sediment, leading to the proverb "rivers are old and lakes are young". However, many lakes in the tropics and large cold temperature water bodies such as Lake Baikal (the world's largest lake) and Lake Ohrid in the Balkans are considered as ancient lakes, many of which are millions of years old. These are the result of large-scale tectonic movements and can host high numbers of endemic species arising from long periods of relative isolation conducive to local speciation, and are well illustrated by the large lakes of the African Rift Valley.

At the other end of the morphological scale from the large ancient lakes are shallow lakes of recent anthropogenic origin. These range from newly water-filled pits following mineral extractions such as for gravel to the shallow lakes of northern Europe that arose from medieval diggings for peat. These are widespread in parts of the Netherlands, Denmark and

East Anglia in the UK (Moss, 1983; Van Liere and Gulati, 1992). Owing to their occurrence in lowland areas with intense human development, many of these shallow lakes are particularly

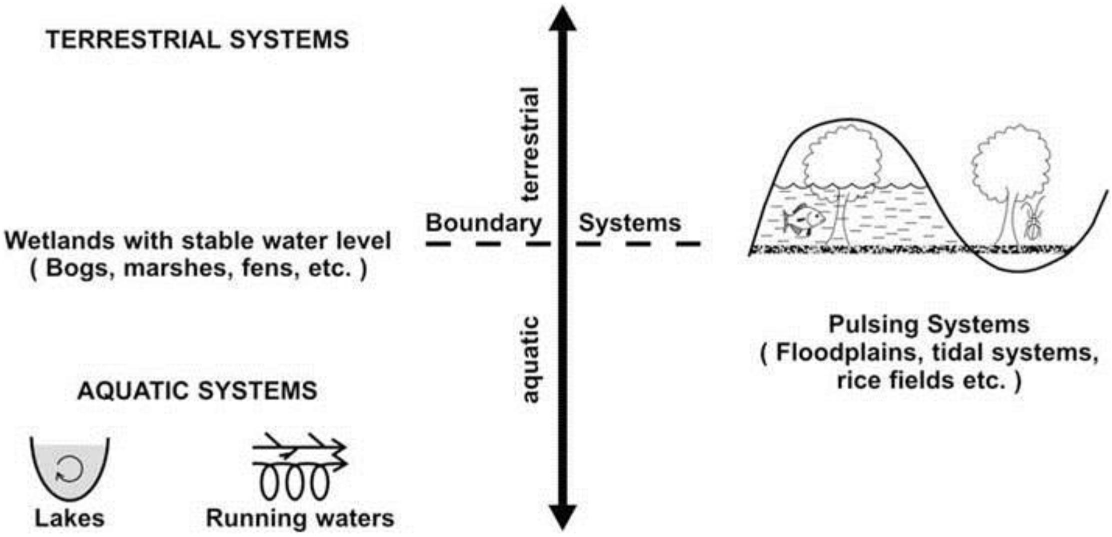
eutrophic from inputs of nutrients from urban settlements and intensive agriculture. Shallow lakes eventually fill in with sediment from the accumulation of internal biological production and material being washed in from the catchment. Although there have been attempts to distinguish small shallow lakes from ponds, this is largely a trivial debate, as these water bodies lie along morphological continua with a variety of attempts at definition (Biggs et al., 2005). Similarly, lake-like expansions of water within river systems have features of both rivers and lakes, often with central channels of low water residence time, and bays where water exchange is relatively low.

Artificially created reservoirs have many similarities in structure and function to lakes, but tend to have more regulated residence times and thermal and littoral structures affected by water drawdown and water release. Reservoirs, particularly those enriched with nutrients and in highly disturbed catchments most likely have an eutrophication problem (e.g. with water hyacinth blooms, and, in severe cases, even cyanotoxins, and also may fill rapidly with sediments, limiting their lifespan. As reservoirs impede river connectivity, they can have important impacts on both upstream and downstream ecological communities through alterations in their flow, sediment and thermal regimes, and the restriction in migration of fish and other biota.

Wetlands

Under the Ramsar Convention on wetlands, wetlands are broadly defined as any land area that is saturated or flooded with water, either seasonally or permanently: “Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres” (http://www.ramsar.org/; see the Ramsar Library, Ramsar Fact Sheet 1: Wetlands: Why should I care? and Ramsar Fact Sheet 6: The Ramsar Convention: What’s it all about?).

Figure 2.3: Classification of wetland systems in relation to other surface waters.
Source: Junk (1989)



This typology of wetlands reflects the continua aquatic systems display, from running water systems (such as rivers and lakes within their basin landscapes) to wetlands with characteristics reflecting more immediate terrestrial influences (e.g. seasonally flooded grasslands), as well as from stable water level systems (such as many lakes) to pulsing inland or coastal ones (e.g. river floodplains and estuaries) (»Figure 2.3).

Wetlands therefore encompass an immense diversity and complexity of types of systems, within three basic categories:

- Inland wetlands, which include aquifers, lakes, rivers, streams, marshes, peatlands, ponds, flood plains and swamps
- Coastal wetlands, which include all coastlines, mangroves, saltwater marshes, estuaries, lagoons, seagrass meadows and even coral reefs.
- Human-made or artificial wetlands, which include fish ponds, ricefields, salt pans (and stormwater retention basins

In the basic typology presented in this report (»Section 2.2) a simplified wetland classification system is provided. Within this more basic typology, vegetation-dominated wetlands, such as Phragmites and Cyperus swamps associated with riverine riparian corridors and lacustrine fringing habitats are included under rivers and lakes respectively. Similarly, for practical purposes, water-dependent upland peat bogs and valley bottom fens, and other types of palustrine wetlands are represented under the “Other permanent/temporary standing water” category (»Figure 2.2). These various wetland types are covered in more detail by the guidance and supporting, hierarchically based wetland typology provided by the Ramsar Convention (www.ramsar.org), as well as by some country or regional inventories, where these exist. The different wetland types could be incorporated in national guidelines for water quality, where wetlands are considered priority systems and where there are sufficient data for assessment.

For wetlands, a hydrogeomorphic approach (HGM approach) has been developed assessing wetland functions (Smith, 1995). Based on geomorphic position and hydrologic characteristics, seven different wetland classes were identified (Brinson, 1993): Depressional wetlands[14], wetlands, mineral flats, organic flats[15], tidal flats, lacustrine fringe and slope wetlands. Groundwater and surface waters are intricately intertwined with, for example, groundwater contributing to the base flow of rivers. While groundwater bodies have recently been considered as ecosystems, their biodiversity is not well known (see »Section 2.1.3).

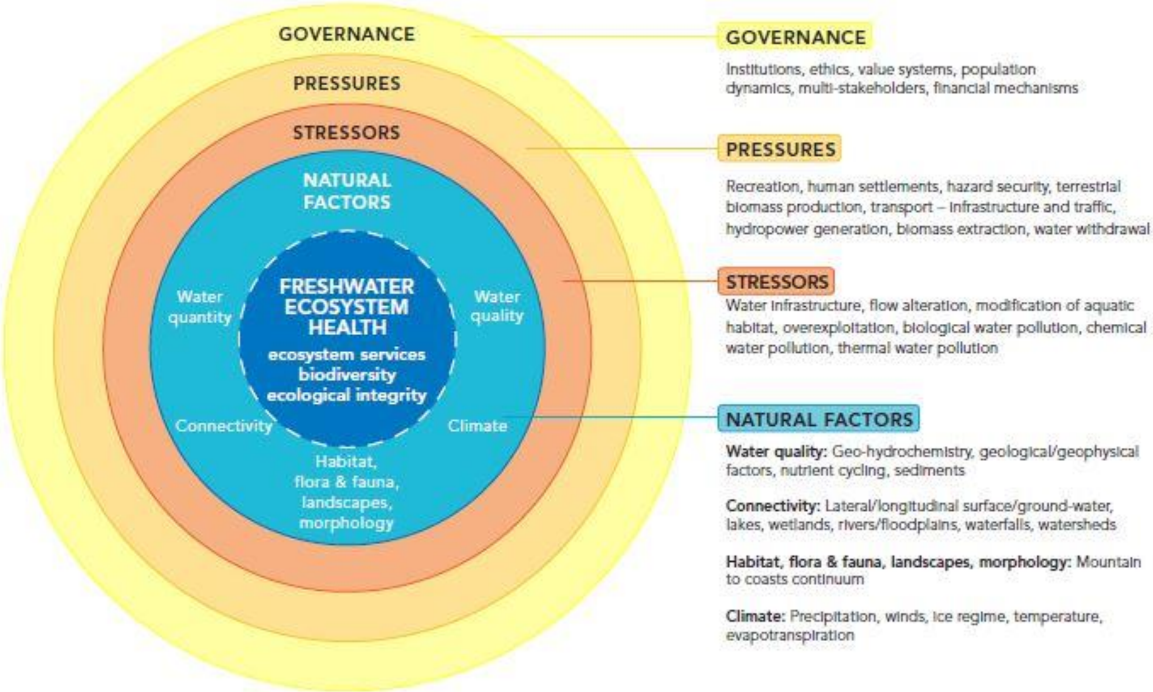
Groundwater or aquifer ecosystems are considered separately, but included in the proposed typology, as they may represent a dominant or priority water body type in certain regions. To date, supporting quantitative guideline information on indicators for groundwater systems and other wetland types is scant, but the proposed classification framework provides placeholders for potential future development of content for these other important water bodies. The interaction with groundwaters can be of major importance for the ecological character of surface waters. Groundwater inputs can affect basic properties such as water temperature, alkalinity and residence time of a surface water body, and provide a conduit for pollutants and aquatic organisms.

Considering the above-listed complex characteristics of the numerous kinds of inland water bodies, it was agreed that at this stage the Framework will focus on surface inland water bodies (lakes, reservoirs and running waters) and the related transitional freshwater-dominated ecosystems such as deltas, estuaries and palustrine wetlands. Groundwater will be considered only in terms of natural outflows into surface water systems or through its links with the water-use cycle, whereby groundwater is abstracted first for utilitarian purposes and discharged into surface waters after these uses.

The conceptual diagram (»Figure 2.4) illustrates the interconnections among freshwater ecosystem health, biodiversity and services, and ecological attributes (e.g. water quantity, stressors, pressures and supporting governance structures).

[14]http://el.erdc.usace.army.mil/wetlands/depressional.html
[15] http://el.erdc.usace.army.mil/wetlands/organic.html

Figure 2.4: Conceptual framework: Freshwater ecosystem health in focus.



Overview of typologies and adopted classifications

Responsible governmental organizations must manage large landscapes composed of an immense density and diversity of freshwater ecosystem types, often with a minimum set of data. Therefore, systematic approaches are required in order to cope with this complexity and to deal with the scarcity of available data and information. Indeed, many approaches to typify freshwater bodies have been developed during the past decades. The assumption is that geomorphology provides a starting point to evaluate the interaction of biophysical processes within a catchment, since it determines the structure of a river system. The structure and function of many rivers are tied to vegetation cover and composition. Also, they influence the diversity of habitats and other facets of aquatic ecosystem functioning. For all these reasons, the rehabilitation programme of a riverine ecosystem should be started with the reconstruction of its morphology and vegetation associations.

However, the successful integration of typology models into ecosystem management faces two main challenges (Soranno et al., 2010): Firstly, there is a need for predictive models, which use variables that allow the establishment of causal relationships between the typology and response of the ecosystem to certain measures; secondly, ecosystem typology must be explicitly linked to management goals. In the present case, the management goals are based on the assessment of the state of ecosystem health and include key principles as the “non-deterioration principle”, the “restoration principle” (defined in the European Water Framework Directive - EU WFD; EC, 2000a) and the “antidegradation” (defined in the US Water Quality Standards Handbook[16]). Hence, the key management aims are

- to maintain high-quality ecosystems in the long-term, thus not accepting deterioration compared to the present state and
- to improve the ecological condition of currently degraded ecosystems, thus improving the state of those ecosystems which have historically been the subject of deterioration.

By applying a landscape approach, a hierarchical or nested ecosystem typology scheme provides the basic framework for ecosystem typology. A landscape approach explicitly

[16] http://water.epa.gov/scitech/swguidance/standards/handbook/chapter04.cfm

integrates the different types of freshwaters (i.e. lakes, rivers and wetlands are not treated in isolation), the heterogeneity of a landscape at different scales as well as the interactions with the surrounding terrestrial matrix. Therefore, this approach integrates variables of the freshwater, terrestrial and human landscape. Tree diagrams allow ecosystem types to be classified in a hierarchical way. However, it is most unlikely that a single typology model meets management goals for multiple response variables. As already mentioned above, it is crucial that typology models are closely linked to management aims. Therefore, it has been recognized by researchers and managers that there is a need for:

- an explicit statement of goals that are linked to quantifiable objectives,
- consideration of multiple spatial scales as drivers of ecosystem dynamics,
- consideration of the hierarchical organization of freshwater ecosystems,
- a foundation of mappable data that is available for all ecosystems and
- the use of monitoring in an adaptive fashion (Higgins et al., 2005; Soranno et al., 2010).

A second approach that needs to be considered is systematic conservation and management planning following the CARE principles: comprehensiveness, adequacy, representativeness and efficiency. Efficiency is usually provided by a complementarity-based strategy, aiming to select new conservation areas in the light of previously protected features (Linke et al., 2011). Systematic planning requires surrogates for conservation assessment (e.g. species surrogates or highly informed physical surrogates, depending on the scale and data available) as well as hierarchical approaches both in scale as well as in detail of assessment. As stressed above, a clear link to the management goals should be established beforehand. Systematic conservation planning has the great potential to identify high-quality areas (such as reference sites) and to set clear priorities for conservation and management. At the same time, it helps to better integrate freshwaters into the terrestrial, coastal and marine realms; i.e. not treating freshwaters in isolation from their interlinked terrestrial, coastal and marine systems (Linke et al., 2011).

Ecosystem health approach principles

Utilitarian and ecosystem oriented approaches

On utilitarian grounds, management is justified only to the extent that it directly benefits human society. In this rather straightforward, anthropocentric concept, everything is considered from the viewpoint of humankind and holds that the proper course of action is the one that maximizes overall human well-being. In other words, the moral worth of an action is determined solely by its resulting outcome (morality can be judged only after knowing all the consequences). Applying to water, ecosystems and the services provided have value to human societies because people derive utility from their use, either directly or indirectly (use values). ES that are currently not used can also be valued by humans (non-use values). Under the utilitarian approach, an effort is made to try to quantify the costs and benefits of different ES.

Utilitarianism faces a number of problems. It is often too difficult to estimate and compare the values of certain costs and benefits, not talking about the future consequences of a course of action. Perhaps, the most serious difficulty of this approach stems from the fact that it fails to consider the role of environmental justice, equity considerations for non-humans and the fact that it may lead to ecosystem degradation, and in consequence even affect utilitarian uses, with a potentially negative legacy for future generations.

In contrast, a non-utilitarian, ecocentric approach recognizes an intrinsic value of all beings, human and non-human, and the benefit of all organisms, with human beings not privileged over other species. Ecocentrists state that no economic gain could outweigh the successful maintenance of intact, undisturbed ecosystems. The approach is based on ethical principles, beliefs and rights, and it is not amenable to the same kind of cost-benefit analysis as utilitarian arguments. From an ecocentrist´s perspective, ecosystems should be preserved and protected with the utmost respect and should not be compromised by the needs of society.

Nowadays a ”pure use value“, utilitarian approach does not cover all societal aspirations anymore, while the ”protect all the species“, ecocentric policy, by ignoring or preferably eliminating human use, is, on the other hand, simply not feasible in reality. Thus, a sustainable compromise solution or balance between the eco- and anthropocentric (utilitarian) approaches is to be sought. It is important to take an ES perspective, and in developing (national) Water Quality Guidelines (WQGs) for ecosystems, the ecosystems themselves need to be more explicitely considered as legitimate water users to ensure their protection and safeguard their freshwater biodiversity. It should be recognized that humans inevitably rely on services which are provided by ecosystems; exhausting freshwater ecosystems is short-sighted and could entirely undermine their service provision. Nevertheless, with more than seven billion people now on earth, often living close to freshwater bodies, it is simply illusory to attempt to keep or re-establish undisturbed ecological status for all water bodies worldwide.

The historical development of water quality evaluation systems has followed the utilitarian approach. Numerous sets of parameters have been developed to classify streams and lakes as potential sources of drinking water or as resources for various other uses. Recent decades have witnessed the emergence of the ecocentric approach. There is much less “consolidation” in this approach than in the “utilitarian” one.

For the purpose of the development of (national) WQGs for ecosystems, a number of practical issues should be considered. In some countries, systematic water quality sampling has about 50 years of well-documented history of continuous development (e.g. of devices, standards, typology and classification schemes). Secondly, the availability of capacity, affordability and willingness to pay play a clear role. Thirdly, the current ecological status of many receiving waters is so poor, that the achievement of a ”good” ecological state (or health) is not realistic in the short term and sometimes not even in the long term. Thus, the advice is to consider phased development within an adaptive management framework: Firstly, improve the physical and chemical water quality, and move towards biological restoration subsequently, as implemented in the EU and the US lately. In other words, a dual approach is taken: The short-term goal is anthropocentric, use-oriented while the longer-term goal or vision acknowledges the importance of healthy ecosystems, among others, for the sake of sustainable human uses. Furthermore, aquatic ecosystems are living entities with their specific water demands. Only after having satisfied these requirements can ecosystems be expected to function and provide ES sustainably. Achieving ecosystem health objectives will likely satisfy many of the other needs as well, while the converse does not necessarily hold true.

Reconciling human use aspirations and ecosystem needs: the ecosystem health concept

Ecosystem health is an integrative field that brings together the biophysical understanding of how near-natural systems function with societal goals and human values (Rapport et al., 1998). Ecosystem health represents a desired endpoint of environmental management. A healthy ecosystem has the ability to maintain its structure and function over time in the face of external stress (Costanza and Mageau, 1999). The human dimension is a central theme in which humans value the goods and services that healthy ecosystems provide for a range of needs and uses, and where unhealthy ecosystems satisfy only a subset of these. In the case of freshwater systems, these may include essential goods and services such as clean drinking water or fisheries production, but also encompass other important societal goals such as aesthetic and cultural values, and biodiversity conservation (Meyer, 1997; Bunn, 2003).

While traditional approaches to water quality assessment, based primarily on physical and chemical indicators, may be sufficient to ensure that some societal values for freshwater are met (e.g. standards for raw drinking water), these are unlikely to be sufficient to ensure that societal goals of ecological sustainability are being achieved.

The concept of ecosystem health is based on the principle that a healthy ecosystem is one that has intact structure and function. Ecological integrity and near-natural conditions with complex food-web connectivity, smooth adaptation of species, efficient nutrient and energy cycling characterize a state associated with ecosystem health. The precise definition of ecosystem health is, however, problematic and analogies with human health are likely overstated, although of some use for general communication.

Assessment of the ecosystem health of inland surface waters has been most developed for rivers, integrating river hydrology, geomorphology, ecology, social considerations and stakeholder views to evaluate trade-offs in the multiple use of ES and ecosystem conditions.

The ecosystem health approach recognizes and mitigates impact to the river in an attempt to maintain key benefits and services of the river. The ecosystem health approach is ostensibly an integrated approach that can also be used to make important decisions on river basin development and its economic and ecological consequences (King et al., 2008).

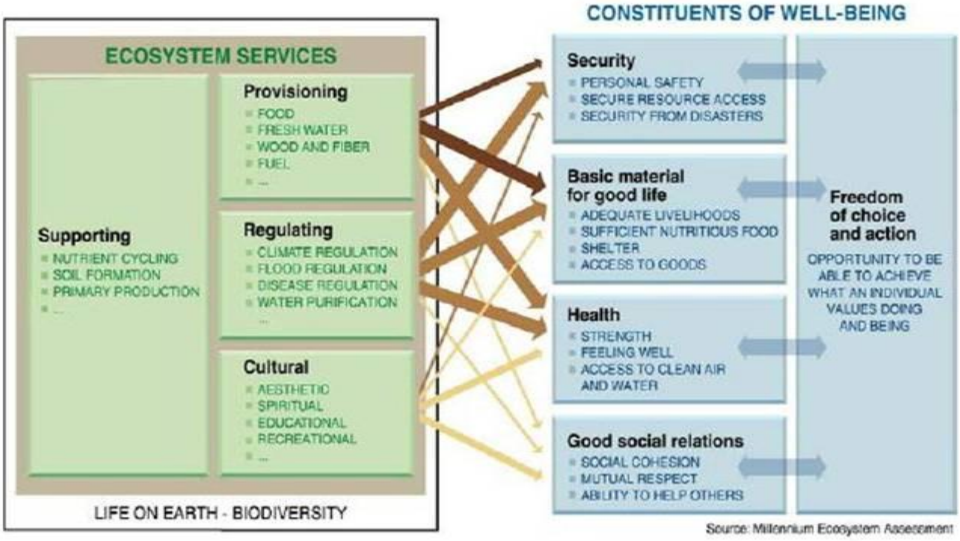
Ecosystems are complex and dynamic. Their functioning is based more on probabilistic than deterministic outcomes and they only attain equilibria within wide bands of possible states. The applicability of the concepts of ecological health are, therefore, more likely to apply to societal agreement on beneficial use (Karr, 1995) than notions of some optimal state (Scrimgeour and Wicklum, 1996). Along a continuum of pristine or minimally impacted high quality sites to those that are clearly degraded, healthy aquatic sites are those that have beneficial use in the ES they provide, but may not be free of anthropogenic pressures that impact their natural structure or function.

Ecosystem services and ecosystem health

The increasing recognition that water bodies and wetlands provide important ES links the natural, social and economic capitals that form the pillars of sustainable development. The economic valuation of water and wetlands illustrates the high economic importance of many of these services, operating on local, regional and global scales (Russi et al., 2013).

»Figure 2.5 (MEA, 2005b) shows the general links between different dimensions of ES and constituents of human well-being. The achievement of these human aspirations is hence based on the sustainable provision of these services. However, »Figure 2.5 does not indicate the feedback loops, i.e. how far the different achievement levels of human well-being create a pressure on the ecosystems. In extreme cases, these pressures cannot be compensated by the natural adaptation capacity of ecosystems and they become stressed. Hence, the various pressures and/or combinations thereof may become a stressor and causes an impact on the state of the ecosystems and could impair the sustainable provision of certain services.

Figure 2.5: Links between ES and human well-being.
Source: Millennium Ecosystem Assessment (MEA, 2005b).



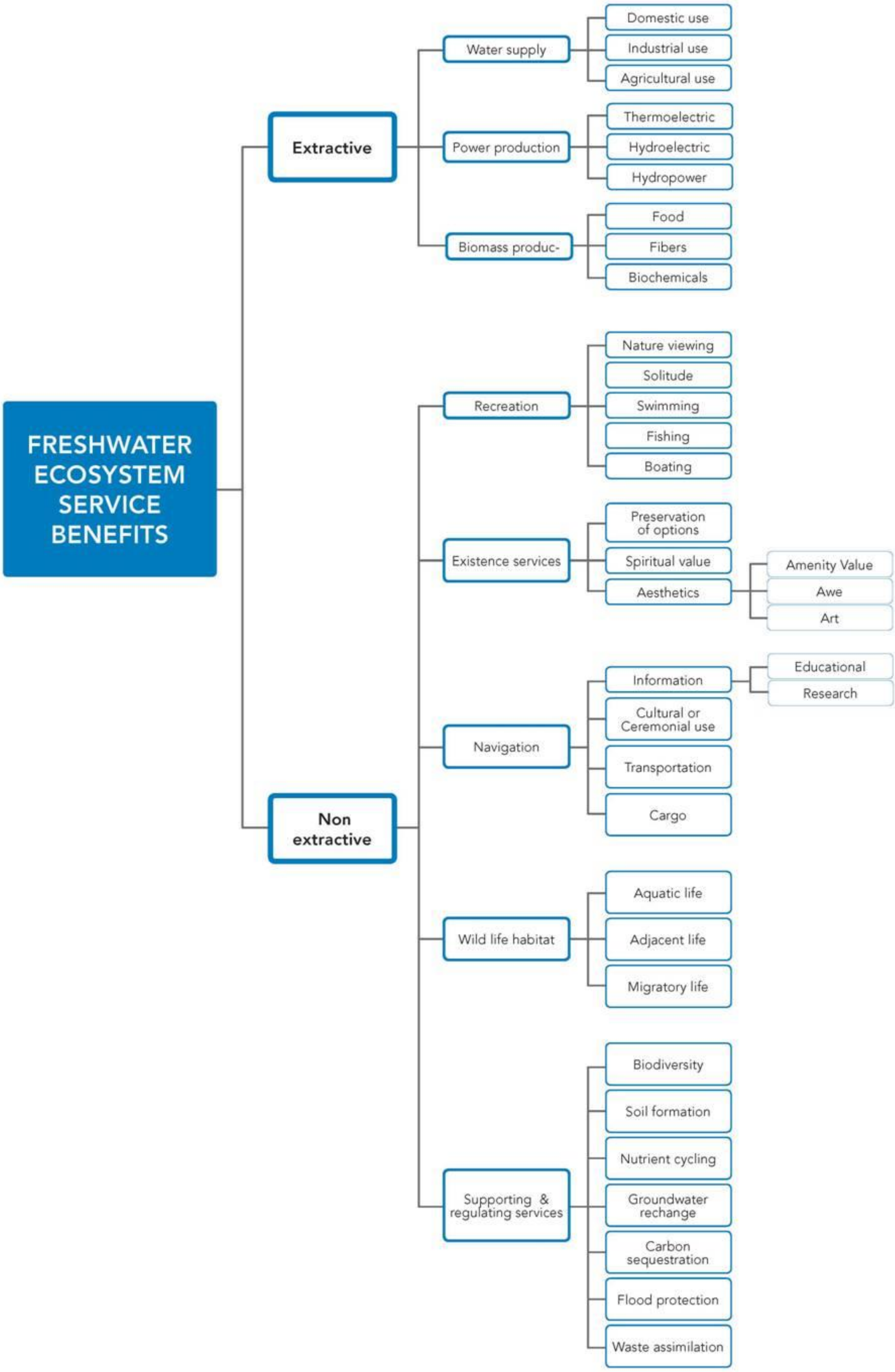
Inland waters provide more ES to humans than open oceans and most terrestrial systems (de Groot et al., 2012). Despite major uncertainties in data and methods, it is well accepted that the rapid loss and degradation of these systems has resulted in an immense loss of their services. Inland waters, for example, play a key role in the global carbon cycle by releasing (through evasion) 2.1 Pg C per year globally (Raymond et al., 2013). Lakes store large amounts of carbon over long periods; in most cases, lake sediments contain more carbon than all soils in their respective drainage area (Cole et al., 2007).

Ecosystem Services (ES) can be classified as

a. Provisioning services,
b. Regulating environmental services,
c. Supporting services and
d. Cultural services (MEA, 2005b).

Estimates by the United States Environmental Protection Agency (US EPA) of the ES from surface waters are shown in »Figure 2.6. This general model applies to lakes, rivers and wetlands, recognizing that each particular site will have its own unique matrix of services, which can be related to ecosystem health. Each relevant component in »Figure 2.6 can be evaluated separately, using a mix of social and natural science, and potentially also economic (monetary) approaches.

Figure 2.6: Typical ecosystem services provided by surface waters
Source: Adapted from US EPA: http://archive.epa.gov/aed/lakesecoservices/web/html/ecosl.html



The ES concept allows evaluation of trade-offs and synergies amongst different services, and allows the quantitative and spatial assessment of the implications of a choice (e.g. MEA, 2005a). Economic methods for the monetary valuation of ES are being developed (EEA, 2010; Puhspam, 2012). Some critics of the ES concept perceive it as equivalent to economic valuation and privatization (Engel and Schäfer, 2013), but valuation should and must not be limited to monetary approaches to make the ES concept operational. Methods of multi-criteria analysis and the combination of quantitative and qualitative approaches can raise awareness about the multiple roles and values of ES (Busch et al., 2012; Chan et al., 2012). This can support deliberative processes and help to identify and negotiate complex trade-offs between different water demands in terms of quality and quantity, including those of aquatic ecosystems (ESAWADI 2011, see also »Annex 5).

- Concepts of quality status, ecosystem health and beneficial use can be aligned into an overall scheme that recognizes and differentiates: the ecological status of the water body (e.g. as described in the EU WFD);
- the ecosystem health approach as epitomized by the e-flow approach (Box 2.2); and beneficial use (e.g. as identified by the US Clean Water Act - CWA).

Combining these three approaches to surface water management leads to gradients along scales of impact and societal trade-offs. It helps to identify where ecological integrity is compromised and what the societal risks and benefits of this impairment might be. The components of this approach provide for the:

- Definition of High Status and Reference Condition (RC) sites as defined by the EU WFD, which maintain the ecological integrity of biological communities through minimal impact, ecological structure and function compatible with efficient cycling of nutrients and energy transfer, and free of invasive alien species. Departure from “High Status” leads to a gradient of successive degradation of the ecological state.
- Assessment of Ecosystem Health that, through a legitimate and transparent governance involving technical expertise and stakeholder engagement, allows for the documented trade-off of ecosystem integrity to satisfy agreed goals of resource use, while mitigating impact to meet the needs of multiple stakeholders and environmental quality compatible with basic standards that safeguard against pollution and hydrological modification that affect a site and downstream use.

The definition of beneficial use is that, on the one hand, it provides for the legal protection of ecosystems of high status and their management, while, on the other hand, it allows compromising and modifying ecological integrity to support socio-economic activities. These impaired ecological zones can, nevertheless, be regulated or managed to mitigate excessive deterioration of status.

One of the principal ways in which ecosystem health and services benefits to people from inland waters have been maintained and/or improved is through water management approaches that incorporate environmental flows (see »Box 2.2).

Box 2.2: Environmental flows as a management tool for people and nature

Environmental flows (or e-flows) are essential for aquatic ecosystem health, the provision of ES to people and human wellbeing. Defining Environmental Flow Requirements (EFR) is a process designed to make well-informed decisions on modified flow regimes as a management response to address the stresses imposed on aquatic ecosystems by hydrological alteration due to water infrastructure (e.g. dams, diversion weirs) and/or water withdrawals for water use sectors such as irrigation, hydropower, industry and municipal use.

Environmental flow is now widely defined by the international community of practitioners as follows:

Environmental flows describe the quantity, timing and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems. (Brisbane Declaration 2007; signed in September 2007, Brisbane, Australia, by over 750 water professionals from more than 50 countries).

The definition acknowledges the principal aspects of a flow or water regime to be considered - quantity, timing and quality. It recognizes the biophysical structural components and processes needed for the healthy functioning of diverse, entire ecosystems and their biota. It also explicitly includes the vital linkages to the livelihoods and well-being of people, particularly those communities that are directly dependent on natural resources for their daily subsistence needs and security.

Several other valid definitions with similar scope and intent exist (see Dyson et al. 2003, for example) and have been adapted in different countries across the world for local use. However, common to all definitions are that: Environmental flows are a pattern of seasonally and annually changing flows and/or water levels that are to be maintained in rivers and other types of water bodies, and not withdrawn for off-stream uses; and that the flows or levels are set to meet a specific set of resource management objectives agreed by society.

Numerous methodologies and tools exist and are continually being adapted for determining the EFR of various types of ecosystems (e.g. river-floodplain systems, estuaries, groundwater-dependent wetlands) and in different degrees of modification from their natural state (e.g. from high conservation value, near-intact systems through to those that are already quite modified for human use) (Tharme 2003). In the case of water bodies that have become heavily modified to meet human needs over an extensive period of time, and for hybrid and novel ecosystems, environmental flows may be most effectively targeted towards achieving specific outcomes such as ES (Acreman et al., 2014). The approaches differ considerably in terms of their strengths and deficiencies in different resource contexts, such as data availability, time, and level of output resolution and expertise required. For further reading on methodology types and examples of their application worldwide, among others, see: Tharme (2003); Dyson et al. (2003); Acreman and Dunbar (2004); Annear et al. (2004); O'Keeffe and Le Quesne (2009); Arthington (2012); Adams (2014). The majority of methods are appropriate on a project scale, but, increasingly, attention is being focused on landscape level, regional or basin approaches to scale up efforts more rapidly in a scientifically robust and defensible way. The Ecological Limits of Hydrologic Alteration, ELOHA (Poff et al. 2010; Arthington, 2012) is one such approach that continues to evolve, including in terms of the social process and associated factors (e.g. Finn and Jackson, 2011) and the context for future water resource management (see discussion and additional reading in Matthews et al., 2014). The ELOHA framework flexibly allows scientists, water-resource managers and stakeholders to analyse and synthesize available scientific information into ecologically based and socially acceptable goals and standards for environmental flow management.

Nested within Integrated Water Resources Management (IWRM), environmental flows are one pragmatic route for the operationalization of IWRM for environmentally and socially sustainable water management. In practice, however, environmental flow implementation, including the adaptively managed allocation of water to the environment through negotiated optimization and tradeoffs with other water users over time, has lagged behind (Le Quesne et al., 2010). To be implemented, environmental flows must be specified in, for instance, basin policies and plans, water allocation plans, impact assessments, water permits or infrastructure operating rules (Hirji and Davies, 2009). For the adaptive inclusion of e-flows in dam operation, for example, Richter and Thomas (2007) and Warner et al. (2014). Le Quesne et al. (2010) identify key factors for successful implementation, based on a series of case study analyses from different countries

Pressures and stressors of freshwater ecosystems

Distinctions and interactions of pressures and stressors

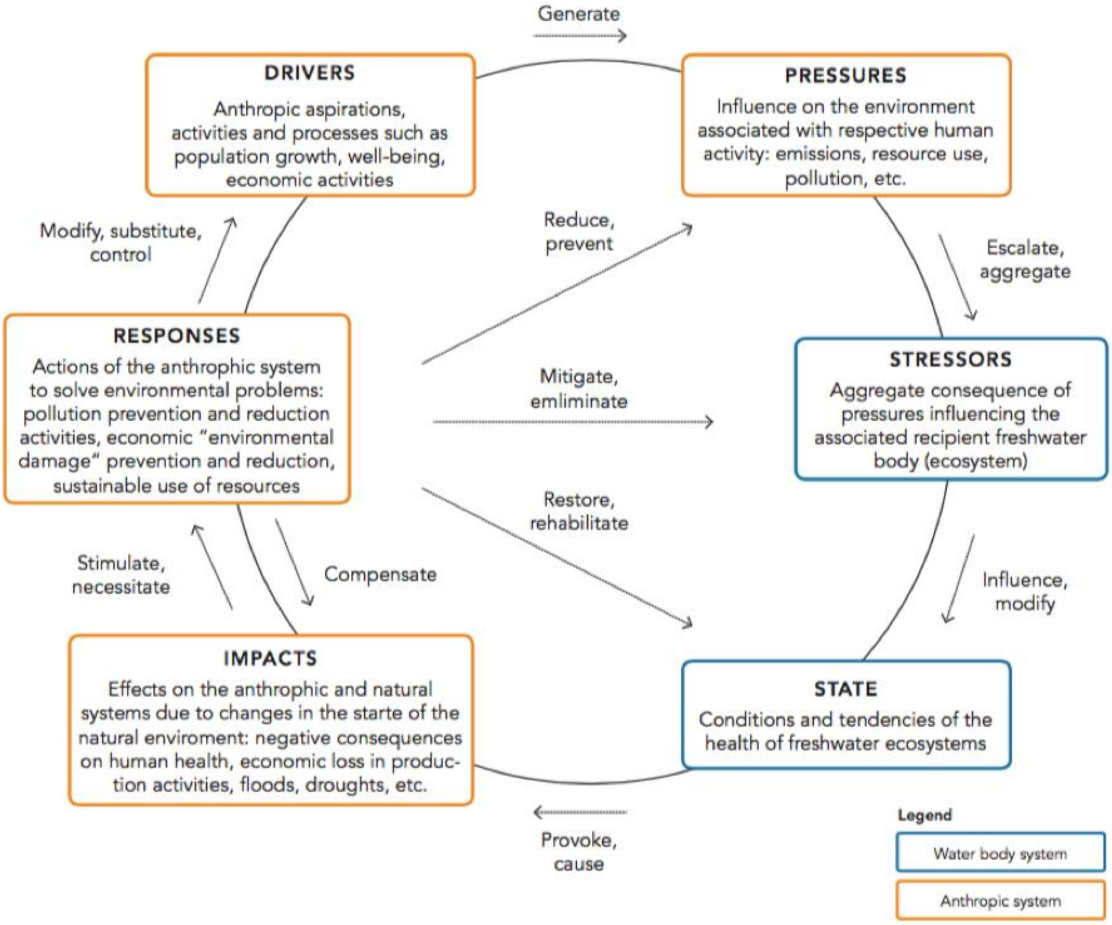
Inland waters are among the most altered systems globally. More than two third of all large rivers are fragmented; reservoirs trap more than 25% of the total sediment load that formerly reached the oceans (Vörösmarty and Sahagian, 2000). Out of the estimated 40,000 km³ annual terrestrial water flux (aggregated stream flow and aquifer outflow to the oceans) (Trenberth et al., 2007) approximately 10% is withdrawn (Rockström et al., 2009). As a global average, around 70% of all water withdrawn is used for agricultural (mainly in irrigation) purposes (Wallace et al., 2003).

Furthermore, more than 50% of the global human population lives within 3 km of a water body; less than 10% of the population lives at a distance greater than 10 km from a water body (Kummu et al., 2011). For example, in Europe and Japan, about 50% of the population currently lives on (former) floodplains (Nakamura et al., 2006). River deltas are, in particular, very fertile areas and are, therefore, among the most populated areas globally (e.g. deltas of the Ganges/Brahmaputra/Meghna, Nile, Rhine, Mekong, Irrawady and Yellow Rivers) (Ericsson et al., 2006).

All life on Earth, including humans, depends upon the integrity of ecosystems for their well-being and survival (Lindenmayer and Likens, 2010a). Aquatic ecosystems are no exception in that they touch all parts of the natural environment and nearly all aspects of human life and culture. Their role in providing natural resources such as fish and clean water are well known as are their role in providing transportation, energy, dilution of pollutants and recreation (Naiman and Bilby, 1998). As a result, very complex inter-relationships between socio-economic factors and near-natural hydrological and ecological conditions have developed. As a consequence of this close relationship, the integrity of aquatic ecosystems is often challenged (Bartram and Balance 1996; US EPA, 2006b). Therefore, freshwaters need to be considered as coupled social-ecological systems where human benefits and ecosystem health cannot be treated in isolation.

Humans not only benefit from the processes and services provided by freshwaters; concurrently, human activities have profoundly altered the physical, chemical, biological and morphological characteristics of inland waters globally. Indeed, most ecosystems are exposed to multiple human-caused pressures. These led in many cases to stresses including water pollution, flow modification, habitat degradation, overexploitation, introductions of alien species which became invasive (Allan and Flecker, 1993; Dudgeon et al., 2006; Malmqvist and Rundle, 2002) (»Figure 2.7). Pressures are the consequence of human activities seeking to satisfy various dimensions of human well-being. Pressures can reach levels beyond the natural resilience of the respective ecosystems. Their functions are impacted and they start to deviate from their ‘healthy state’. Thus, the pressures become sources of stress. Several pressures may aggregate and cause stress or one type of pressure might cause different kinds of stresses, which are directly relevant for inland surface water bodies. Thus, pressures are conceptualized in association with activities for human well-being, whereas stressors can generate negative impact, causing potentially aggregated effects of pressures to the supporting freshwater ecosystems. This distinction was introduced to account for the focus on ecosystem health in this Framework. The proposed modified Drivers, Pressures, States, Impacts, Responses (DPSIR) cycle with the additional “stressor” element (thus Drivers, Pressures, Stressors, States, Impacts, Responses (DPSSIR) model) to emphasize the interface between the anthropic and freshwater ecosystem is shown in »Figure 2.7.

Figure 2.7: Linking the anthropic and freshwater ecosystems and their causal chains of links. Source: Modified based on ISTAT, C. Costantino, F. Falcitelli, A. Femia and A. Tuolini (OECD Workshop Paris, 14-16 May 2003).



A matrix of pressure/stressor relationship is presented in Table 2.1; an example of interactions in »Box 2.3. Pressures such as water withdrawals for domestic, industrial, mining, agricultural and energy generation (cooling water) purposes, and the subsequent discharge of used (waste) waters, but also fisheries, aquacultures as well as sand, gravel and other mineral removal from rivers and lakes constitute extractive and (subsequently) discharge pressures. Hydropower generation and navigation are typically in situ pressures while transport infrastructure, traffic, terrestrial biomass production, urbanization and recreation, but also security requirements against water-related hazards, can be classified as riparian/ basin scale pressures. Climate variability and climate change as well as various aerosols and depositions constitute additional, global-scale pressures through the easy connectivity within the atmosphere.

The diverse pressures and stressors rarely act in isolation; they often have synergistic or additive effects that impede management prioritization. Interactive effects complicate both the prediction and alleviation of impacts. For example, when managers introduced small beetles to control the introduced Eichornia crassipes (water hyacinth) that covered much of the surface of Lake Victoria in the 1990s, they successfully countered one problem, yet the overall outlook for the lake and its species remains bleak because of the ongoing degradation of the lake and its catchment (Hecky et al., 2010). Not only do different types of stresses often have cumulative effects that exceed their individual impacts, but restoration efforts targeting one factor may have small overall effects if other stresses or threats remain or even increase.

Box 2.3: Interaction of stress factors: Controlling water hyacinth in Lake Victoria

Climate change is triggered by activities conceived to increase human well-being. The 5th IPCC Assessment (2013) indicates that direct human impacts such as land use and land use change, water pollution, and water resource development will continue to dominate the threats to most freshwater ecosystems over the next 3 decades (Settele et al., 2014). However, climate change will exacerbate many of these pressures, thus showing how combinations of increasing pressures contribute to worsen several stressors. Rising water temperatures are likely to lead to shifts in freshwater species distributions and worsen water quality problems, especially in those systems with high anthropogenic nutrient loading (Settele et al., 2014).

Land use alteration and its inherent land cover change are the consequence of multiple human activities along shorelines or within the catchment of the respective water body that might impede freshwater ecosystems through increased sedimentation, nutrient enrichment, contaminant pollution or hydrological alteration. Riparian clearing, the loss of large forests and other human activities have shaped the state of terrestrial and freshwater ecosystems globally for thousands of years. Agriculture and deforestation are the dominant land use changes globally. Urban land use, while increasing, typically covers a smaller percentage of catchments, but, due to its disproportionate influence on aquatic ecosystems, its role is nevertheless important. Indeed, land use pattern and human density in the catchment often serve as suitable surrogate indicators of freshwater conditions, acting as a general index of human disturbance (see also Sections 4.3 and 4.4). The proportions of cropland and urban area as well as the state of the riparian zones are probably the most effective proxies reflecting the environmental state of freshwater ecosystems (Bunn et al., 1999; Peterson et al., 2011).

Pressures influence water bodies through different, sometimes indirect links such as water withdrawal, discharge, seepage, atmospheric deposition, rainfall and radiation.

Stressors are understood as concrete and negative manifestations of pressures on inland waters such as construction of water infrastructure (dams, barrages, sluices, ports, dykes, groins or other types of artificial obstacles within water bodies), alteration of flow and water levels (through withdrawals, discharges, backwater effects, hydropower generation and the operation of water infrastructures), modification of aquatic habitats (dredging, mining, river training), overexploitation of aquatic resources, biological water pollution such as the emergence of invasive alien species but also that of pathogens and genetic modifications in freshwater ecosystems, chemical and thermal pollution (mainly through the discharge of wastewater and returning cooling water) (»Table 2.1).

For example, the stressor ‘overexploitation’ depends mainly on pressures emanating from various water uses (withdrawal, wastewater discharge), fishing and aquacultures, dense population (settlements and recreation) and indirectly through climate change as a limiting factor of ecosystem resilience. As such, climate change “alone” could also become a stressor itself. The main stressors of freshwater ecosystems are described in the following sub-sections.

Table 2.1: Pressures and stressors relevant for inland surface waters. Pressures exert their influence on water bodies through hydraulic structures and river training, withdrawals, discharges, seepage through ground water bodies, atmospheric deposition, rainfall and radiation.

Stressors / Pressures	Water Infrastructure (artificial /physical obstacles)	Flow alteration	Modification of aquatic habitat	Overexploitation	Biological water pollution (invasive species, pathogens, etc.)	Chemical water pollution	Thermal water pollution
Water withdrawal/discharge (domestic use)	Strong	Strong	Indirect	Strong	Indirect	Strong	Strong
Water withdrawal/discharge (Industries incl. extractive ind.)	Strong	Strong	Indirect	Strong		Strong	Strong
Water withdrawal/discharge (agricultural use)	Strong	Strong	Strong	Strong		Strong	
Cooling water withdrawal/discharge	Strong	Strong	Indirect			Strong	Strong
Biomass extraction (e.g. fishery and aquaculture)				Strong	Strong	Strong	
Mineral extraction (sand, stones, gravel, gold, etc.)		Strong	Strong	Indirect		Strong	
Hydropower generation	Strong	Strong	Strong			Strong	Indirect
Navigation	Strong	Strong	Strong		Strong	Strong	
Transport infrastructure and traffic	Indirect	Strong	Strong			Strong	Strong
Terrestrial biomass production (food, timber, energy crops, animal husbandry & fish ponds, etc.)	Indirect	Strong	Strong		Indirect	Strong	
Hazard security (flood protection, etc)	Strong	Strong	Strong				
Human settlements (esp. in the proximity of water bodies)	Indirect	Strong	Indirect	Strong		Strong	Strong
Recreation			Strong	Strong	Indirect		
Climate variability and change, atmospheric deposition		Strong	Strong	Strong	Strong	Indirect	Strong

(pink)	Extractive pressures, withdrawals, intruding discharges
(green)	In situ pressures
(yellow)	Riparian/basinwide pressures due to land use and landcover change
(peach)	Atmospheric pressures
(dark blue)	Strong links
(red)	Indirect and secondary links

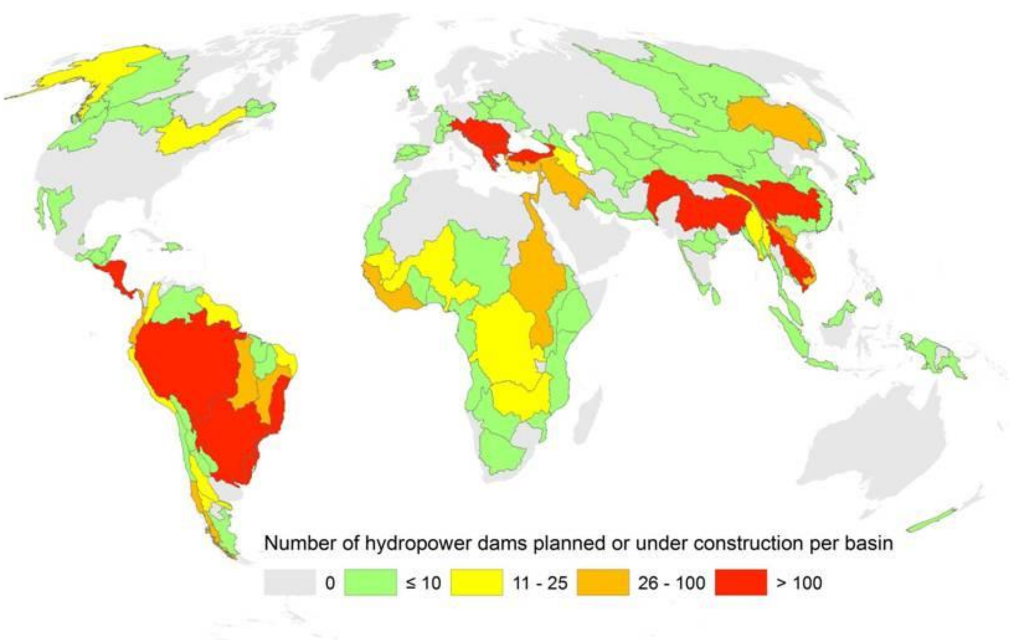
Water infrastructure

Infrastructure development including dams, levees, port and harbour infrastructures, bridges and other engineering structures located in or constraining water bodies are stressors which modify water flow, lateral and longitudinal connectivity and hydromorphology with potentially harmful effects on freshwater species, including fish, molluscs and reptiles, since they often cannot adapt to these changes resulting in increased risk of extinction (Allen et al., 2012; Smith et al., 2014). Possible impacts include:

- Alteration of the hydrology (see »Section 2.4.3). River flow regimes (including long-term average flows, seasonality, low and high flows, and other types of variability) play an important role for freshwater ecosystems;
- Decrease in biodiversity due to river regulation and dredging, shoreline development, and extensive habitat loss;
- Truncation of the longitudinal, lateral and vertical connectivity, which impedes the migration and dispersal of many organisms, in particular of fish, and reduces the transport of material and resources;
- Degradation of wetlands, flood plains and fringing buffer zones from levee construction or inundation by impoundments.

Although dams were mainly built in the developed countries in previous decades, the trend to plan and build dams and hydropower facilities has clearly shifted as demonstrated in »Figure 2.8. Given the growth in human population, economic development and the rapid rise in energy demand, this stressor is most likely gaining in importance. Indeed, within the next 10-20 years, hydroelectricity production will almost double, thus shaping the global river network considerably.

Figure 2.8: Number of future hydropower dams per major river basin. Source: Zarfl et al. (2015).



Flow alteration

Hydrology is considered the “master variable” in inland waters (Jackson, 2006; Poff et al., 1997). Therefore, disruptions of the hydrologic regime alter natural processes, ES and biodiversity. Hydrological alteration may be defined as “any anthropogenic disruption to the magnitude or timing of near-natural stream flows” (Rosenberg et al., 2000). Such changes in the magnitude and pattern of flows (or water levels), caused by the storage, regulation, diversion and/or extraction of surface and groundwater by dams and other water resources infrastructure, are one of the primary contributors – often the major one – to the degradation in riverine ecosystem structure and functioning, and decline of freshwater habitats (Postel and Richter, 2003). The physical (hard) and so-called soft (e.g. altered thermal regime) barriers created by water resources infrastructure fragment aquatic systems, blocking species movements between habitats and during migration, disconnecting rivers from their floodplains and associated wetlands, and changing temperature, nutrient and sediment gradients and other processes (e.g. delta building) needed for life cycle activities such as fish spawning and vegetation recruitment. The water resources infrastructure involved is associated with the development, reliable delivery and use of water for irrigated agriculture, energy production, flood protection, and supply to communities and businesses. In addition to such effects due to water management, climate change, and its adaptation responses, such as increased water storage, may also profoundly impact the hydrological, thermal, nutrient and sediment regimes vital for ecosystems and biodiversity.

The construction of dams has already profoundly altered the character and condition of rivers and other wetland ecosystems. By the end of the 20th Century, over 58,000 large dams (dam height: > 15m) had been constructed globally across more than 140 countries[17] (WCD, 2000).

About 20-25% of continental runoff and about 25-30% of the total global sediment flux in rivers are now held behind reservoirs (Vörösmarty et al., 2004; Vörösmarty et al., 2003). Global fragmentation of rivers by such hydrological alteration is well documented. Nilsson et al. (2005) showed that 59% of the world’s large river systems (accounting for 60% of world runoff) were fragmented by flow regulation and channel fragmentation associated with dams, including the world’s eight most biogeographically diverse systems. More recently, Lehner et al. (2011) report that 50% of the length of all rivers with discharge > 1000 m3 s-1 is impacted. They estimate that about 16.7 million reservoirs larger than 0.01 ha – with a combined storage capacity of approximately 8070 km3 – may exist worldwide, increasing Earth's terrestrial surface water area by more than 305 000 km2. Some 65% of continental discharge is considered under moderate to high threat in terms of human water security and biodiversity (Vorosmarty et al., 2010). Further major hydrological alteration, with its attendant impacts on wetland ecosystems, is on the horizon, as highlighted for the hydropower sector alone in Zarfl et al. 2015 and Opperman et al. 2015.

By substantially changing the natural patterns of river flow and blocking the movements of fish and other biota, large dams, in particular, have severely disrupted the natural food production systems of rivers (e.g. fisheries, flood-recession agriculture). This diminished food security has placed large downstream populations and their livelihoods at considerable risk (Richter et al., 2010).

There are about 3,700 major hydropower dams either planned or under construction. These dams, if implemented, may almost double the total installed capacity from hydropower from currently 900 GW to more than 1600 GW (Zarfl et al., 2015). Hot spots of future dam construction include South America, Southeast Asia, including the Himalayas, Africa, the Balkans, Anatolia and the Caucasus regions. Many of the basins where such development is planned are also significant in terms of their conservation

[17]http://www.icold-cigb.org/GB/World_register/general_synthesis.asp

assets and ES values (Opperman et al., 2015). Development of scientifically sound standards to decide on location, type and operation of future dams and hydroelectric power plants would be helpful.

More than half of all rivers globally are temporary (»Section 2.1.2), meaning that they fall dry at the surface for given periods of time; more permanent rivers are expected to turn temporary in the future due to climate change and overexploitation. The transformation of permanent to temporary waters fundamentally alters biodiversity and ecosystem processes. Flow intermittency per se is not necessarily a stressor in water courses having this phenomenon as a natural feature; however, human-caused alteration of flow regimes is frequently associated with other stressors such as water pollution and species invasion (Acuna et al., 2014). Additionally, ecosystems might not be adapted to these typically fast changes of flow.

Climate change induced changes in precipitation will substantially alter ecologically important attributes of flow regimes in many rivers and other wetlands, and increase impacts from human water use in developed river basins (Döll and Bunn, 2014). Around the world, changes to flow regimes resulting from shifts in precipitation and evaporation patterns have already been documented (Rosenzweig et al., 2007).

Through environmental flow assessments (see Box 2.2), scientists and water managers are now seeking to define the patterns of flow in a river, lake or other water body that can sustain it in a healthy condition under such future development scenarios.

Modification (degradation) of aquatic habitat

Habitat degradation is a universal stressor on all inland waters. For example, more than 50% of all wetlands have been lost worldwide (Finlayson and D'Cruz, 2005). Large-scale losses of habitat are expected to continue, particularly in the developing world, as inland water systems are further modified to provide electricity, water for irrigation, drinking water and sanitation services.
Changes in land cover increase sedimentation, enrich nutrients, alter flow and lead to a decline of riparian areas (Allan, 2004). Clearing of natural land cover for agriculture and other land uses can have similar impacts on the hydrology, although the interactions are complex and impacts are place-specific. In rivers, increased erosion following deforestation and other land use change can lead to inputs of sediment that decrease light penetration, clog the bottom habitat and disrupt the overall functioning of the ecosystem. In small Amazonian streams, clearing of tropical forest and conversion to pasture has been shown to change the biogeochemical and hydraulic characteristics of the system (Neill et al., 2006). At the extreme, whole mountaintops are removed for mining operations and the resulting dredge material is disposed of in nearby valleys, burying entire streams (Palmer et al., 2010).
More subtle degradation of aquatic environments is also commonplace. For instance, removal of wood debris from streams and lakeshores facilitates navigation and human recreation, but at the cost of simplifying the habitats. This can adversely affect populations of fish and other aquatic organisms.

Overexploitation

Overexploitation refers to both overstressing water bodies in their function to provide ES (such as fishing or absorption of water pollution), but also to excessive withdrawals and extraction of mineral resources. Overexploitation may affect ecological processes and biodiversity including evolutionary processes. Although it is extremely difficult to determine the status of inland fisheries because of underreported catch data, there are strong indications that inland fisheries in most parts of the world are heavily exploited

(Dugan et al., 2007; Kura et al., 2004). In Europe, for example, more than 20 million recreational anglers use and exploit inland waters. In many systems, we are “fishing down the food chain” – as larger species become overexploited and rarely caught, smaller species make up the bulk of the fishery. Some examples are stated in »Box 2.4.

I. Freshwater turtles: Disappearing species

The situation for freshwater turtles is similarly dire. The demand for turtle species for traditional Chinese medicine and food supply in Southeast Asia has decimated regional turtle stocks. As of 2000, annual exports from Indonesia, Bangladesh, Thailand, Malaysia and Taiwan, amounted to 15,500 metric tons, representing approximately 10.3 million average market-sized adult turtles annually, or 28,300 turtles per day. As demand continues to increase, harvest has expanded to parts of North and South America (Turtle Conservation Fund 2002).

II. Overfishing - a global threat

The catch of the Oueme River fishery in West Africa, for example, was composed of large species reaching about 60 cm in length in the 1950s; by the 1990s, the length of species caught had been reduced to 10-30 cm (Allan et al., 2005). Evidence exists of similar declines in the largest species in other fisheries in the tropics, as in parts of the Amazon and Mekong Basins (Allan et al., 2005; Castello et al., 2009). Large species of commercial value in the northern hemisphere, such as sturgeon and salmon, are also heavily exploited. A recent review concluded that sturgeon and paddlefish are imperilled across the globe and long-term survival of these species in the wild is in jeopardy (Pikitch et al., 2005).

Box 2.4: Examples of overexploitation

Biological water pollution

Biological water pollution covers invasive alien species (also known as exotic, introduced or non-native species) and refers to species, subspecies or lower taxa occurring outside of the range they occupy naturally or could not occupy without direct or indirect introduction or care by humans. Although the majority of alien species cause no harm, some alien species spread very rapidly and can harm biological diversity, human health, and/or economic and aesthetic values. These harmful species are called invasive (alien) species[18] (see »Box 2.5 for an example).

Primary vectors for the introduction of aquatic species include deliberate introductions, aquaculture escapees, inter-basin water transfers, ballast water from vessels, canals, and releases from aquaria, gardens and bait buckets (Strayer, 2010). Deliberate introductions occur for a variety of reasons - primary among these is the commercial or recreational harvest of the introduced species and biological control of other introduced species.

Species invasion may lead to faunal homogenization, alter ecosystem processes and, in some cases, cause the extinction of native species (Olden et al., 2008; Rahel, 2000). Beyond invasive species, the category of biological water pollution includes the occurrence of pathogens and parasites, threatening humans (Conn, 2014) and aquatic species (Ashander et al., 2012; Meyer et al., 2016; Spikmans et al., 2015), and effects on the genetics of native species through escapes of captive bred stocks of e.g. fish (Baskett et at al., 2013).

[18] See http://www.europe-aliens.org

Box 2.5: Massive financial losses: Cases of invasive species worldwide

In East Africa's Lake Victoria, for example, a complex set of threats - the introduction of Nile perch along with overfishing, climate change and habitat degradation due to land use change in the basin – have jointly caused a decline in abundance and loss of a large proportion of the 500 native cichlid fish that once inhabited its waters (Chapman et al., 2008; Hecky et al., 2010). The eutrophication of the lake has stimulated extensive growth of water hyacinth (Eichornia crassipes), an invasive species native to South America. The plant, which forms dense patches on the water surface, is now widespread across Africa, with often major impacts on navigation, depletion of water column oxygen, and replacement of other floating and submerged aquatic plants. The explosion of zebra mussels in Western Europe and North America, for example, has caused the decline of some species and significantly altered habitat conditions for others. The economic impacts exceed US $100 million per year in the US alone (Strayer, 2009). Overall, it is estimated that damage and losses due to invasive species in the United States add up to almost $120 billion per year (Pimentel et al., 2005). Water hyacinth is a major weed in South Africa, where it alters the water regime. More than US$ 25 million per year is spent on its control, additionally US$ 15 million is spent per year to remedy damage from water lettuce (Huntley, 1996).

Chemical water pollution

Freshwater ecosystems suffer from the input of both nutrients and toxic chemicals due to human activities. Nutrient loading occurs as a consequence of transforming land cover from natural vegetation to farm fields, roads and cities; it may moreover stem from human waste and untreated human wastewater. Most modern agriculture involves the application of large amounts of nitrogen and phosphorus fertilizers in order to enhance crop growth. A portion of these nutrients run off to rivers and lakes, where they can cause overgrowth of both plankton and aquatic plants. In the Chesapeake Bay (US), high nutrient inputs from upstream and atmospheric deposition have resulted in large algal blooms, creating dangerously low oxygen concentrations across large portions of the estuary. Emissions from cars, power plants and industry also contribute to nutrient loading. These emissions disperse in the atmosphere and long-distance atmospheric transport of nutrients has elevated inputs of nitrogen even in remote freshwaters that appear pristine. Near population centres, phosphorus from wastewater is a problem that requires societal investments in proper treatment technologies and control of inputs.

Harmful chemicals are also a widespread threat to human and natural uses of freshwaters. Contaminants such as pesticides, heavy metals, pharmaceuticals and organics can reduce water quality to the point where rivers and lakes can no longer support a full complement of species. For instance, acid rain arising from emissions of sulphur and nitrogen oxides was an acute problem in lakes and rivers of eastern North America and Europe until emissions controls became obligatory (Malmqvist and Rundle, 2002). Highly acidic run-off continues to be problematic downstream of abandoned mine sites, making these streams uninhabitable for most species. A growing list of man-made chemicals used in industry and home products have been found in aquatic ecosystems, and scientists are still struggling to understand their prevalence and impact. Some of these disrupt the endocrine system of freshwater animals and people (Jobling et al., 1998; Mills and Chichester, 2005); for instance, intersex fish possessing both male and female characteristics have been found in all nine of the large river basins sampled in the United States (Hinck et al., 2009). Much work remains to be done in order to be more certain about the consequences of even low concentrations of industrial chemicals that occur in many freshwaters.

Water quality is, moreover, expected to decline in some basins due to higher pollutant loads from heavy precipitation events, overflow of wastewater treatment plants during extreme rainfall and greater volume of withdrawal from low quality sources

(Kundzewicz et al., 2008).
In recent decades, net-cage aquaculture has become one of the main patterns of the intensive fish culture in the lakes/reservoirs in several countries (i.e. Indonesia, China, Ethiopia and the Philippines; »Box 2.6). Net-cage aquaculture is considered one of the major stressors on lake water quality. Organic and nutrient loading from the excess feed and fish waste to the lakes has resulted in organic accumulation in the sediment and lake water quality deterioration and accelerated the process of lake eutrophication and toxic cyanobacterial bloom (Guo and Li, 2003; Hallare et al., 2009; Dagefu et al., 2011; LIPI, 2001). Frequent fish kills due to oxygen depletion in hypereutrophic lakes and/or other factors such as raised levels of ammonia and hydrogen sulphide have been reported in Indonesia (LIPI, 2001).

With the increasing world population, the need for foods such as animal protein is also increasing. Fish can be a source of animal protein that is relatively cheap and easy to obtain. To meet the needs of the fish, one of fish farming technologies known as very profitable, low cost and high production in a relatively short period is floating net cage aquaculture. Cage aquaculture is culture of fish in floating net cages with high fish stocking density commonly applied in an open access water body such as lakes and reservoirs with relatively low water current. Cage aquaculture has been widespread around the world and become one of the major methods for intense fish production in tropical countries with big populations such as in Indonesia and Brazil, but also in African countries and even in China (Henny, 2014; Zhou et al., 2011, David et al., 2015, Gondwe et al., 2011). In Indonesia, lakes and reservoirs are open access water bodies that are considered common property, and therefore with no clear regulation of cage aquaculture. This activity has been intensively growing and is uncontrollable. With the rapid development of cage aquaculture in lakes and reservoirs, its negative effects on water quality and aquatic life structure is clearly emerging. Cage aquaculture has been a major pressure and stressor on lakes' ecosystem quality, as it causes large inputs of nutrient and organic carbon in lakes. A large amount of organic waste from residual fish pellets and fish feces is entering lakes/reservoirs, directly causing nutrient enrichment that leads to excessive eutrophication and even intensive macrophyte coverage in lakes (Guo and Li, 2003; Garg and Garg, 2002, Henny and Nomosatryo, 2012; http://danau.limnologi.lipi.go.id). The amounts of nitrogen and phosphorus that went directly to a reservoir in China through the residual bait and fish droppings in cage aquaculture grew from 42 kg to 768 kg per year and from 10 kg to 856 kg per year respectively during the period from 2007 to 2009 (Zhou et al., 2011). The main source of nutrient input from cage aquaculture in Lake Maninjau, Indonesia resulted in a total of N and P of around 400 tonnes per year (http://danau.limnologi.lipi.go.id).

Fish feeding residue entering the water with net-cages in the Niushanhu Lake, Yangtze Basin, China during the period from March to December 2000 in total covering an area of 1,000 m² was equivalent to 1,532.9 kg of total N and 339.2 kg of total P (Guo and Lie, 2003). High inputs of organic waste from cage aquaculture over the years in Lake Maninjau, Indonesia has also caused the acccumulation of organic matter in the lake sediment and increased high sulphide production in the hypolimnion layer and led to oxygen depletion in the upper water layer. The frequent fish kill has been reported several times over the years (Henny, 2009, Henny, 2014). The eutrophication problem caused by cage aquaculture in lakes is usually accompanied by excessive bloom of cyanobacteria known as the toxic algae of Microcystis. Microcystis population was more than 40% and up to 60% in almost all areas especially near the fish cages in Lake Maninjau (Sulastri, 2012). Prolonged to poor water quality such as being exposed to lower oxygen levels could affect the ecosystem structure in lakes (Hackenson, 2005, Macuiane et al., 2015; Zhou et al., 2011)

Box 2.5: Cage aquaculture practices – facts and figures

Thermal water pollution

Temperature is a key environmental factor, as it influences the biology of every organism. Most aquatic organisms are adapted to a specific temperature range, outside of which temperatures become stressful and ultimately lethal. For example, the optimal temperature range for rainbow trout is between 13–15 °C, with the lethal maximum of 24.3 °C (Bear et al., 2007).

Thermal water pollution refers to an artificial increase or decrease in the temperature of a water body as a result of human activities (Kennedy, 2004). Although enhanced water temperature can have beneficial aspects; altered water temperature and temperature regimes more often have a negative and long-lasting effect on freshwater ecosystems. Effects include lethal or sub-lethal effects of individual organisms and their development, adult migration, competition with non-native species (Riis et al., 2012), and the relative risk and severity of disease (Karvonen et al., 2010). Temperature also influences the capacity of water to hold Dissolved Oxygen (DO), which again affects aquatic organisms in various ways (Kennedy, 2004). Specifically in temperate lakes, thermal water pollution during winter was shown to be stored in the deep water column until the next winter. Accordingly, winter thermal water pollution can have a long-lasting negative effect on lake ecology (Kirillin et al., 2013).

Thermal water pollution (in the sense of the artificial temperature increase of recipient water bodies) is strongly associated with cooling water discharge, first and foremost from various types of power plants. Given the expected growth of energy demand on a global scale, thermal pollution will increasingly become a concern. As the temperature of water bodies should not exceed certain thresholds to remain supportive for aquatic life, it is frequently the case that power plants need to shut down or curtail their power generation during summer periods as well as in the light of the climate change driven increase in water temperature (van Vliet et al., 2012).

A less common form of thermal water pollution involves the release of cold water from reservoirs into warmer receiving water bodies. This occurs, for example, in Australia when cold water from reservoirs is released for irrigation purposes. If the water is released from the bottom of the reservoir, it can be considerably colder than the water in the receiving water body. The effects of cold-water pollution can be similar to that of warm-water pollution, but it has no negative impact on the water's DO holding capacity (Kennedy, 2004).

Climate change induced air temperature shifts are altering surface water temperatures in many temperate lakes resulting in reduced periods of ice formation and the earlier onset and increased duration and stability of the thermocline during summer (Winder and Schindler, 2004). These changes are projected to favour a shift in dominance to smaller phytoplankton and cyanobacteria (Settele et al., 2014). There is widespread evidence of rising temperatures (caused, at least partially by climate change) in streams and rivers over the past few decades, and this has been linked to shifts in invertebrate and fish community composition. These phenomena indicate how closely the different stressors are intertwined. See »Box 2.7 for further information.

Geothermal energy becomes available as a result of the transmission of heat accumulated in the interior of the Earth's crust to the water in an aquifer and the moving of this warmer water to the surface through wells. The water that moved to the surface from underground is used for purposes such as electricity, housing and greenhouse heating, thermal treatment and tourism.

The geothermal water whose energy is used becomes wastewater to a large extent. Geothermal wastewater is generally rich in view of metal content and has a high temperature (for Turkey, for example, it is between 30 and 287 °C). One way to ensure the final disposal of such wastewater is to discharge it to the nearest surface water body. It is known that discharging wastewater without treatment and without any control causes some serious ecological problems. Temperatures of geothermal wastewater are higher than those in the receiving body. Therefore, this geothermal wastewater causes thermal water pollution in the receiving body. Thermal water pollution has two effects: the deterioration of the oxygen balance and the deterioration of ecology. On the other hand, geothermal wastewater, which includes a large amount of metal, causes chemical pollution in the receiving body that ends up giving rise to health problems and the deterioration of ecology. Geothermal wastewater discharging into a receiving body without treatment and without any control causes pollution in the receiving body and this pollution eventually moves into the groundwater associated with the surface water.

Arsenic (As), Boron (B), Fluoride (F) and Silicon (Si) are elements that are often encountered in geothermal water. According to the World Health Organization (WHO), As and F, if taken in excess by drinking water, are the elements that have a negative impact on health. Therefore, the World Health Organization has determined the upper limit of As and F for drinking water at 0.01 and 1.5 mg/l, respectively. For B, the upper limit of 2.4mg/l is proposed. However, in large parts of the world, the upper limit of 0.5 mg/l is used for B.

Another disposal method of geothermal wastewater is reinjection. Reinjection, which can be defined as pumping the geothermal water back into the reservoir after using the energy of the geothermal water for various purposes, contributes to sustainable use by providing additional supply of the geothermal field that is only fed to a limited extent from natural sources. It mitigates the risks of collapse and pressure drops in the geothermal field. In addition, it increases production capacity by providing more thermal energy from the reservoir rock. On the other hand, reinjection can lead to problems such as cooling water in the production well. Furthermore, leaks that can be generated as a result of improper operation or accidents that may occur during the design or operation to these systems can cause serious groundwater pollution which is difficult to discern and treat. In this context, reinjection systems should be designed very carefully in order to provide a sustainable use of the geothermal field and reduce the possible negative effects on ecology.

In summary, countries that use geothermal sources for heating and health purposes are faced with both groundwater and surface water pollution problems. Wastewater that was used for healing and heating purposes must be treated before discharging to the receiving bodies. Geothermal wastewater used for energy should be returned to the same formation where this geothermal water was abstracted.

Information stated in this box is based on personal communication with Dr. Yakup Karaaslan (Ministry of Forestry and Water Affairs/ Directorate General of Water Management, Turkey).

Box 2.7: Geothermal Water Pollution

Responding to the threats

A sustained, global response is required to halt the ongoing losses of freshwater species and the degradation of freshwater ecosystem health. As described in Sections »2.1.3 and »2.3.3, there is the risk of losing many services provided by aquatic ecosystems as well as the richness of biodiversity that they support. In response, society must devise strategies that leverage scientific understanding to reduce threats in ways that both protect aquatic biodiversity and enhance human well-being.

The actions needed to counter these threats are often quite obvious. For instance, requiring adequate flow below dams or the complete removal of dams are relatively simple solutions to the suite of problems arising from damming rivers. However, resource limitations and human needs limit the range of feasible approaches, making it imperative to prioritize actions. Science-based, systematic methods for conservation and restoration planning applied to freshwater ecosystems at national and regional levels have advanced greatly in recent years (Nel et al. 2009). However, further work is needed, particularly to guide prioritization at continental and global levels. Large-scale datasets on species, ecosystems, drivers and threats (i.e., Freshwater Ecoregions of the World[19], DIVERSITAS[20], BioFresh[21], GEOBON[22] and the IUCN Red List of Threatened Species[23]) are already helping to make this goal achievable.

In responding to threats, narrowly focusing on what has changed in the last few hundred years and simply trying to reverse these changes is unlikely to be productive or possible. This is particularly pertinent in planning responses to climate change because it has the potential to completely change the context within which near-natural systems currently operate in the coming decades. Responding effectively to climate change in the context of freshwater conservation requires a shift in the human perception of natural systems and the actions that must be taken to conserve them. For example, as species' ranges shift due to a changing climate, it might be necessary for newly arriving species not to be classified as 'non-native and possibly invasive species', but as native species adjusting to a changing planet. New approaches to 'climate-aware' water management are required in many basins across the globe as are governance structures with sufficient capacity and authority to deliver that management (Matthews et al., 2009). Flexibility and adaptability, also in human endeavours, will be needed, as water managers will have to deal with ever greater climatic and eco-hydrological uncertainty (Matthews and Wickel, 2009; Milly et al., 2008).

Addressing risks and uncertainties

Basic limitations and their consequences

"The simplicity of nature is not to be measured by that of our conceptions. Infinitely varied in its effects, nature is simple only in its causes, and its economy consists in producing a great number of phenomena, often very complicated, by means of a small number of general laws" (Pierre-Simon Laplace in his Philosophical Essay on Probabilities, 1825).

Variability and change are basic features of freshwater ecosystems. The variability that is inherent in nature presents fundamental challenges for monitoring and ecological assessment (Clarke and Hering, 2006), often accentuated by unrealistic demands for certainty by stakeholders and policy makers (Westervelt, 2001). Variation occurs in time and space, on both small and large scales. Even in climatic zones with fairly constant temperatures, natural background chemical concentrations are subject to

[19] http://www.worldwildlife.org/pages/freshwater-ecoregions-of-the-world--2
[20] http://www.diversitas-international.org/
[21] http://project.freshwaterbiodiversity.eu/
[22] http://geobon.org/
[23] http://www.iucnredlist.org/

variation driven by changes in hydrology and the ambient environment. Daily alterations

in oxygen concentrations in water bodies affect redox potential and the mobilization of nutrients and heavy metals, for example. In urban areas, daily patterns of human life affect emissions of pollutants to water bodies. Larger temporal scale variation is subject to the annual rhythm of farming, for example. Floods and droughts can cause dramatic changes in water chemistry and the habitats of plants and animals. Changes in rainfall can drive biological dynamics and human exploitation of natural resources. The need to account for variation is a necessary component of the assessment of water quality, quantity and associated ecology. However, estimating and communicating uncertainty in ecological classification schemes remains a major challenge for environmental policy (Pahl-Wostl, 2002, 2007; Harremoës, 2003; Sigel et al., 2010).

Consequently, collecting data from ecosystems is subject to “errors”, owing to both natural variation and sampling procedures. Even with the high precision of measurements and good quality assurance, inherent variation remains. Natural cycles within species, interactions between populations, food chain relationships, competition, disease and other factors give rise to variation on the scale of years, decades and longer. Biotic response to individual stressors is frequently non-linear and may interact in complex ways across multiple stressors. Introduced (non-native) species, for example, can accentuate or mitigate the manifestation of nutrient enrichment (Byström et al., 1998).

A single sample from a water body only provides a “snap-shot” of information on water quality, but provides no information on where that sample lies within the real distribution of site condition or how close it is to the mean condition. Additionally, the frequency distribution of samples over time and space may be skewed, requiring standardization to enable the application of classic statistics to estimate the mean and variance of the sampled population of data. Uncertainty is reduced through replicate samples, but multiple samples from the same location can lead to pseudo replication, well captured by Bailey et al. (2004, page 31): “We must leave behind the notion that points or areas sampled within a site are in any sense replicates”. Samples replicated over time are subject to seasonally driven variation. Interpreting water quality monitoring results necessitates careful consideration of the design of monitoring programmes and subsequent analysis. Trend analysis of long-term time series overcomes some of these problems, which are inherently and generally complex. The intensity of sampling to reduce errors to an acceptable degree (e.g. < 20%) may be great and far beyond the scope of most monitoring programmes (Fairweather, 1991) if a highly statistical probability of classification of a water body is required. For instance, for Gaussian distributions, the error of a sample mean (e.g. annual average of a water quality component or a tributary emission to a water body) is proportionated to the variance to the mean ratio, while it is inversely related to the sample size. Similarly, small trend per variance ratio could result into so high sampling frequency which would lead to correlated population (Somlyódy et al. 1986) being in contradiction to statistical assumptions of trend detection. As can be seen, variances play an important role to handle uncertainties and for this reason their order of magnitude estimates should be considered from the very beginning of the monitoring design.

The extent of statistical probability that is acceptable should be determined a priori (Downes et al., 2002), and reporting this enhances the transparency of classification schemes. Inbuilt quality assurance protocols reduce field and operational error. Incorporating spatial and temporal variation is more challenging. Nevertheless, a realization of the potential effect of uncertainty in assessment, monitoring and classification guides realistic expectations of information.

Within water classification, misclassifications comprise either: 1) a ‘false positive’ (Type I error) of detecting an impact when there is none or 2) a ‘false negative’ (Type II error) of failing to detect an impact when there is one. Improved knowledge of the statistical distribution of the data used for assessment reduces both the risk and consequences of misclassification. Errors can occur at all stages of a sampling programme, including among operators in field sampling and taxonomy (Prygiel et al., 2002; Furse et al., 1995; Jones et al., 2007), low precision of chemical analysis and the use of difficult-to-

measure metrics and their combination into multi-metrics. The use of a number of metrics combined into a multimetric (such as aggregating indicators into indices) also introduces some mathematical artefacts, as by chance alone the greater number of metrics used in a multi-metric assessment, the greater is the statistical probability of misclassification. Further reading e.g.: Kilgour et al., 1998; Van Sickle, 2010.

Precautionary principle and uncertainty

The precautionary principle states that where a policy or action (including the option of inaction) provides a potential risk of harm to the public or the environment, those causing this risk need to provide evidence that the intended action or policy is not harmful. Scientific (or other) uncertainty or a potentially high degree of risk should not be used as a reason to postpone preventive measures. The precautionary principle is of particular importance in countries with economies in transition and which face severe environmental and health problems (O'Riordan, 2004). In such situations, socio-economic priorities and/or weak enforcement of regulations may compromise protecting the environment and public health. The use of the precautionary principle can, however, inform decisions under uncertainty, highlight the importance for research and innovation and help build public confidence. It can be a safeguard against repeating many of the mistakes made in industrialized countries and attribute stronger responsibilities to those creating the risks.

Guiding principles in adopting a precautionary approach identified by Harremoës et al., (2002) included:

- Acknowledging and responding to ignorance, uncertainty and risk;
- Ensuring that real world conditions are adequately accounted for in regulatory appraisal;
- Ensuring the use of 'lay' and local knowledge, as well as relevant specialist expertise, and taking full account of the assumptions and values of different social groups;
- Maintaining the regulatory independence of interested parties while retaining an inclusive approach to information and opinion gathering;
- Identifying and reducing institutional obstacles to learning and action; and
- Avoiding 'paralysis by analysis' by acting to reduce potential harm when there are reasonable grounds for concern.

While the precautionary approach is mentioned in many policy documents addressing water quality and the environment, it is often not clearly manifested in the actual policy or in practice. An attempt to incorporate the principle into industrial practice is evident in the EU Regulation on Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) (»Box 2.8).

REACH is the Regulation on Registration, Evaluation, Authorisation and Restriction of Chemicals. It entered into force on 1st June 2007. It streamlines and improves the former legislative framework on chemicals of the EU. The main aims of REACH are to ensure a high level of protection of human health and the environment from the risks that can be posed by chemicals, the promotion of alternative test methods, and free circulation of substances on the internal market, and enhancing competitiveness and innovation. REACH makes industry responsible for assessing and managing the risks posed by chemicals and providing appropriate safety information to their users. In parallel, the EU can take additional measures on highly dangerous substances where there is a need for complementing action at EU level.

Source: http://ec.europa.eu/enterprise/sectors/chemicals/reach/index_en.htm

Box 2.8: Example of the precautionary principle in public policy – new EU legislation REACH

Uncertainty in the assessment of the physico-chemical properties of water

Concentrations of chemical variables in surface waters are affected among others by seasonal patterns of climate, hydromorphology, land use and biotic transformations. Chemical measurements are susceptible to intermittent loadings from point or diffuse sources. Inputs from industrial processes and wastewater treatment plants (WWTPs) may be subject to daily variation. Concentrations of sediment and phosphorus loads from land to water may be highly influenced by the hydrologic regime, rainfall events and antecedent moisture conditions (Sharpley et al., 1994; McDowell and Trudgill, 2000). The majority of nutrients and sediments frequently enter rivers during a small percentage of rainfall events and infrequent sampling may lead to large underestimates of total loads (Cassidy and Jordan, 2011). High intensity or continuous monitoring is only possible in a very few selected rivers. Nutrient and other pollutants entering standing waters may be driven by both external sources and, particularly in shallow lakes, internal release from nutrient–rich sediments (Søndergaard et al., 1999; Gibson et al., 2001; Somlyódy and van Straten, 1986).

Even though the chemical state in standing waters is generally less prone to variation than rivers, large seasonal effects can occur when sampling surrogate measurements of the nutrient state, e.g. using phytoplankton Chlorophyll-a (Chl-a) concentration. In shallow lakes, particulate nutrient fractions may already demonstrate hourly alterations due to wind-induced re-suspension (Somlyódy and van Straten, 1986). Seasonal variations may also occur in both temperate and tropical lakes related to patterns of wind, rain and land use.

Uncertainty in the assessment of the biotic conditions of water bodies

Using biotic measures of water quality overcomes some temporal uncertainty, as biota tends to better amalgamate responses to pressures over the medium to long-term: In effect, they are continuously monitoring the water they live in and many species are sensitive to pollution (Resh and Jackson, 1993). However, biotic distributions are often highly influenced by habitat heterogeneity along spatial gradients or life-cycle patterns along temporal ones (Austin and Smith, 1989; Johnson, 1998; King et al., 2000). The variation in biotic community structure varies among regions (Johnson, 1998; 2000; Johnson et al., 2004), water bodies (Tolonen et al., 2001; White and Irvine, 2003), seasons and years (Hamalainen et al., 2003). Statistically robust sampling requires an understanding of the effect of sample and site variation (Stoffels et al., 2005) and the choice of indicators which are not very susceptible to these factors in order to guide sample frequency and intensity.

The original use of biotic indicators in freshwaters was related to the response of river macroinvertebrates to oxygen deficits downstream of sewerage treatment works or industrial outfalls with high organic loads (Hellawell, 1986). In comparatively recent times, the development of classification techniques (Wright, 2000) has been widened to also facilitate the monitoring of the response of river macroinvertebrates and algae to more general nutrient enrichment (Guilpart et al., 2012, Evan-White et al., 2013, Schneider et al., 2013). In contrast to rivers, ecological assessment of lakes has focused mainly on the response of open-water phytoplankton (usually measured as concentrations of Chlorophyll-a) to nutrient (mainly phosphorus) enrichment (OECD, 1982) and, to a lesser extent, that of profound or sub-littoral invertebrate communities (Thienemann, 1931; Naumann, 1931; Saether, 1979; Lauritsen et al., 1985; Lang and Reymond, 1993, 1996; Dinsmore et al., 1999).

In recent decades, there have been increasing attempts to develop classification techniques in both lakes and rivers based on the response of a range of biota to both chemical and physical pressures and stressors (e.g. Haury et al., 2006; Schaumburg et al., 2004; Fjellheim and Raddum, 1990; Wiederholm and Johnson, 1997; SWEPA, 2000; Milbrink, 1978; Reynoldson et al., 1997; Wolfram et al., 2002; Karr et al., 1986). Coinciding with these efforts has been attention to uncertainty in assessment methods (Clark, 2003; Johnson et al., 2006; Walley and Fontama, 1998a,b) including the effects of sample size (including sub-sampling procedures) within site replication, habitat heterogeneity and taxonomic resolution. To comprehensively consider these and related issues, the US EPA (2013b) released a review that illustrates all these processes, including materials, that can be used to evaluate the technical rigour and breadth of capabilities of a biological assessment programme. The goal of such reviews is to identify the strengths and weaknesses of assessment programmes, and to identify areas of potential enhancement. It should, however, be noted that relationships between direct measures of pressures and stressors and water quality metrics can be highly significant, but of low predictive ability (Håkanson, 1999). Moreover, variation in impacted sites can be quite different from that in unimpacted ones (Donohue et al., 2009).

Uncertainty in the assessment of toxicants and multiple stresses

The response of ecosystems and component organisms is a function of critical loads of impact (Nilssson and Grennfelt, 1988). For toxic chemicals, these loads are often determined by laboratory toxicological testing of model organisms, often invertebrates or fish, with subsequent extrapolation to field conditions. Uncertainty in this process, or limitations of data, is often accounted for by precautionary safety margin factors (of up to three orders of magnitude) for permissible (i.e. “no effect”) field concentrations. Such margins of safety are used when there is high uncertainty of an effect, although lower margins are used when there have been extensive trials of chronic toxicity (ECB 2003). When data on the effects of potential toxicants are limited while modelling, so-called “Quantitative Structure Activity Relationships” can be used to estimate acute or chronic toxicity, or persistence and bioaccumulation. They can be used to set environmental limits (Furusjö, 2006). Such models have been developed for numerous organic complexes, but not, for example, for individual metals. Increasingly, there is a need to account for multiple stresses, having combined or synergistic effects (Ormerod et al., 2010). Combining toxicants with nutrient enrichment can, for example, lead to a variety of complex responses and a wide range of impacts (Aragón-Noriega and Calderon-Aguilera, 2000; O’Toole and Irvine, 2006; SCHER/SCCS/SCENIHR, 2012). The use of specific trait based assessment that incorporates relative sensitivity and life cycle traits (von der Ohe & Liess, 2004; Liess et al., 2008) has been used to classify species at risk, but developing effective techniques and models for assessing impact from multiple stressors will increasingly be important for water quality monitoring (Altenburger et al., 2015).

A particular category of multiple stressor is when physical alterations to water bodies affect habitat structure, and water flow and retention. Such hydromorphological alterations may not only affect habitat and species compositions, but a myriad of responses to other pressures. The prevalence of such impacts is global and increasing. They include alteration of river flows through impoundments, weirs, channelization and river basin transfers, physical modifications of lake shores and estuaries, and changing hydrological connectivity between water bodies and catchments. Each effect changes estimates of uncertainty in water monitoring, although may also lead to much less uncertainty where habitats are simplified through engineering. In such circumstances alterations in hydromorphology can lead to a more consistent, or regulated, flow or water retention regime, while in others can result in quite dramatic upstream, or downstream effects.

Uncertainty in ecological assessment

As ecological variables are inherently continuous rather than categorical, fixed boundaries in ecological assessment introduce considerable uncertainty for classification schemes (Carstensen, 2007; Hering et al., 2010). The closer a metric score is to a class boundary, the greater the statistical probability of misclassification (Clarke, 2000; Ellis, 2006). Estimating probabilities around class designation (Clarke, 2010) provides a more realistic, and in the longer term, useful approach for management that is dependent on fixed-boundary schemes. How to apply these in data-poor situations remains challenging and considerable further work is required to link uncertainty to boundary conditions and subsequent dissemination to water managers (Carstensen, 2007; Carstensen and Henriksen, 2009; Hering et al., 2010). Appreciating the uncertainty of such boundaries in ecological assessment and classification (Irvine, 2012) can liberate managers from hopeful expectation and over-reliance on such boundaries, and help incorporate uncertainty into realistic assessment procedures. Such procedures should include general principles of management geared towards a philosophy of no-deterioration and continuous reduction of risks to water quality achieved through catchment management planning, and commensurate monitoring and reporting.

Spatial and temporal variations, although always important, are more readily addressed in monitoring or research programmes dedicated to individual sites rather than those across geographical and temporal scales. Robust site-specific assessment requires long-term monitoring or robust site-specific models (Clarke et al., 2003; Pont et al., 2006; Aroviita et al., 2009; Carvalho et al., 2009; Jeppesen et al., 2005). While this may be a logical solution to dealing with uncertainty, it is not feasible for most water bodies. It can be effectively employed in sites that have a high cultural or biodiversity profile, are economically important or for those sites that can serve as regional “ecological barometers”.

Regional monitoring often ignores spatial and temporal variation in the interests of expediency, but failing to appreciate uncertainty in nature can lead to the misclassification of sites and, consequently, misguided management. Moreover, the timing of seasonal maxima in water bodies can be difficult to predict and vary markedly across water bodies, even in close proximity to each other. Variation in data owing to spatial and seasonal patterns can be reduced through sampling well-defined habitats (White and Irvine, 2003; Tolonen et al., 2001). Habitat stratification, whereby sites are sampled according to their physical similarities, reduces internal noise in the data and increases power for detecting change among sites (Johnson, 1998; Johnson et al., 2004; Resh and Jackson, 1993; Parsons and Norris, 1996; Pinel-Alloul et al., 1996). For routine assessments, inherent variation should, as a matter of course, be minimized. However, sampling “particular habitats” loses information on overall biodiversity. If that is the goal of sampling, then integrated sampling of multiple habitats may be more appropriate, accepting the increase in uncertainty in relating community structures to anthropogenic pressures and stressors.

In conclusion, natural variation is an inherent feature of ecosystems. As uncertainty in site classification is always present, the appreciation of its possible consequences for water management is a fundamental component of water quality assessment. Detailed site-specific assessment, often coupled with local understanding of system functioning, always provides better information to guide management than regional sampling or the use of global models. However, as regional water quality assessment can seldom benefit from such a detailed approach, a more general one is necessary. This can be supported by models that predict the link between pressures, stressors and impact, targeted monitoring of short-term events, and long-term time series to guide understanding. Uncertainty in water quality assessment is reduced by clear aims for monitoring and quality assurance at all stages of the process, from sampling to reporting. Communicating uncertainty to stakeholders and water managers ensures transparency and makes for better decision making.

Dealing with the uncertainty inherent in ecological assessment requires a management approach that is both adaptable and able to learn from experience.

Resilience

In connection with ecosystem health classification and transitions between domains, the resilience of the respective system plays an essential role. Resilience has several definitions. In ecology, it stands for the robustness of the system to remain in its "original" state, irrespective of stresses impacting the system (Holling, 1973). In general systems theory, resilience is defined as the ability of the system to bounce back to its "original" state, irrespective of whether it temporarily deteriorated into a "lower status" state. In this context, resilience is measured by the time needed for recovery (Hashimoto et al., 1982). Furthermore, resilience is increasingly characterized as "the capacity of a system to absorb disturbance and reorganize while undergoing change so as to still retain essentially the same function, structure, identity, and feedbacks" (Walker et al., 2004). This definition is the one considered in the proposed Framework. Linked to this concept is the notion of thresholds or tipping points, which can be defined as "a breakpoint between two regimes of a system" (Walker and Meyer, 2004). This Framework therefore looks at resilience as a measure of the likelihood of a tipping point (irrevocable shift of the ecosystem health from a higher into a lower category) being passed.

Thresholds of water quality and ecosystem health, as defined in this Framework, do not necessarily constitute tipping points in ecological systems, although the threshold between the highly modified/polluted class and the benchmark state of the 'dead zone' could be considered as a tipping point, especially under consideration of the corresponding utilitarian and ecosystem service dimension of this transgression. Obviously, being close to a category limit increases the chance of sliding below it. Resilience depends on how far away an ecosystem is from the RC and how far a system is destabilized though this relationship is unlikely to be linear. Biological indicators (e.g. groups and traits of species, redundancy in traits, trophic/functional indicators, etc.) are especially capable of comprehensively providing information on the status of an ecosystem. Consequently, they may give the best estimate for its resilience.

Key elements of monitoring

Rationale for monitoring

Water quality monitoring for freshwater ecosystems can either be question driven or mandated, which is reflected in the different available definitions of monitoring such as monitoring "…is a time series of measurements of physical, chemical, and/or biological [and hydromorphological] variables, designed to answer questions about environmental change" (Lovett et al., 2007, page 253) or as defined by the International Organization for Standardization (ISO) as a "programmed process of sampling, measurement and subsequent recording or signalling, or both, of various characteristics, often with the aim of assessing conformity to specified objectives".

In more detail, monitoring can have various rationales such as

- to draw and follow up on cause-effect relationships by correlating system responses to external events, stressors or various conditions,
- to show or forecast the direction, speed and magnitude of a historical or potential change,
- to detect and forecast limits of the system (tipping points),
- to measure the effectiveness of the programme of measures,
- to verify whether the freshwater ecosystem water quality is suitable for intended uses,
- to control and warn if preset values or thresholds are exceeded, but also
- to support evidence-based decision making.
- Beyond these explicitly technical rationales, monitoring is essential to provide the evidence base triggering policy actions, strategical planning and management interventions.

The questions which monitoring programmes seek to answer can vary greatly in scale (both time and space) and level of resolution. Meybeck et al. (1996) differentiated monitoring activities into the categories of monitoring, survey and surveillance where monitoring was seen as

i. a 'long-term, standardized measurement and observation of the aquatic environment in order to define status and trends' as opposed to
ii. surveys with finite duration carried out for a specific purpose and surveillance, which was seen mainly as mandated, continuous measurement for the purpose of water quality management and operational activities.

The same authors in the same chapter of the cited book also discussed and emphasized the need for careful and appropriate setting of objectives in order to achieve a successful monitoring and assessment programme. More recently, Lindenmayer and Likens (2010a) categorized monitoring programmes into the categories of curiosity-driven or passive, mandated, and question-driven. In many if not most cases, a given programme will not fit neatly into any single category, especially in cases where monitoring programmes are designed to meet multiple programmatic needs. This does not negate the need to carefully define programmatic expectations at the outset of programme development activities (US EPA, 2006b, Lindenmayer and Likens, 2009, 2010b).

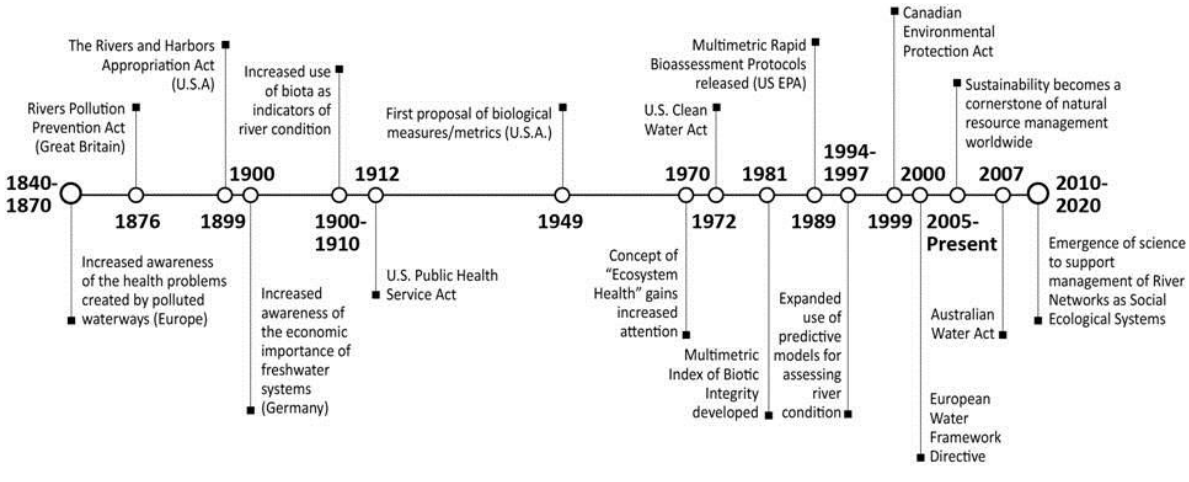
Monitoring and assessment are also crucial elements of the adaptive management process (Walters, 1986) and must be clearly linked to identified values and objectives. Assessment implies analysis and interpretation of data from monitoring and related research studies, and is informed by a conceptual understanding of likely cause and effect and other lines of evidence (see Downes et al., 2002). The aim of assessment is to synthesize monitoring findings and identify potential management actions. While most monitoring programmes are focused on a condition, state and trend, assessments can also include information on pressures and stressors (e.g. Vörösmarty et al., 2010) or the effectiveness of management actions (Lindenmayer and Likens, 2010a).

Clearly comprehensive and accurate assessment of trends in water quality that raise awareness of the urgent need to address the consequences of present and future threats of contamination and to provide a basis for action at all levels is very much necessary. Reliable monitoring data is the indispensable basis for such assessments (Bartram and Balance, 1996). Ecological information collected as part of monitoring programmes helps to meet this need by providing valuable insights into status and trends in ecosystem structure, ecological processes and the services freshwater ecosystems provide. Without this information, there is only limited knowledge about the changing status of the life support system of the planet (Lindenmayer and Likens, 2010a). Appropriate legal frameworks and legal mechanisms as policy means to produce action towards monitoring and the collation of monitoring results is however a prerequisite and a key element of monitoring.

A brief history of assessment and monitoring of aquatic ecosystems

The development of aquatic ecosystem assessment and monitoring has been described in a number of publications (e.g. Cairns and Pratt, 1993; Bonada et al., 2006; Friberg et al., 2011; Parsons et al., 2016). Population growth and its increased concentration within urban areas since the industrial revolution has resulted in increasing amounts of effluents being discharged into local water courses and lakes (Bonada et al., 2006). Health risks resulting from these exposures led to the development of bacteriological methods to monitor the concentrations and impacts of effluents (Bonada et al., 2006). The turn of the 20th century saw the emergence of the use of biological organisms such as plants, macroinvertebrates and fish in monitoring programmes (Kolkwitz and Marsson, 1909). Programmes have continued to evolve in content and approach, and now range in complexity from the least sophisticated programmes that may focus exclusively on a single element (e.g. water quality/chemistry) to integrated assessment programmes that monitor a suite of elements (e.g. water chemistry, physical habitat and biological assemblages). Parsons et al. (2016) presented a timeline of important developments in assessment and monitoring since the mid-19th century (»Figure 2.9).

Figure 2.9: Timeline of important developments in river assessment and monitoring since the mid-19th century. Source: Parsons et al. (2016).



Water chemistry indicators provide direct measures of water quality and are often associated with legislated water quality standards. Common parameters measured in the field include DO, conductivity, turbidity, pH and temperature (albeit the latter not being a chemical measure). Those measured in the laboratory can include common indicators such as nutrients (e.g. total phosphorus and nitrogen) and simple cations and anions (e.g. sulphate and chloride).

A physical habitat assessment examines the structural features of aquatic ecosystems. Habitat and biological diversity are linked, and the loss or damage of habitat is one of the principal stressors to biota (Raven et al., 1998). Recent approaches to the physical assessment of rivers have adopted a hydromorphological perspective, which emphasizes that the interaction between the flow of water and channel form is key to river condition (Newson and Large, 2006; Vaughan et al., 2009; Vogel, 2011; Elosegi and Sabater, 2013).

Biological assemblages are the central focus of many assessment and monitoring programmes as they provide a direct measure of biological condition relative to biological integrity. Biota integrate the effects of multiple stressors in space and time (Rosenberg and Resh, 1993) and thus provide a way of detecting stressors that may be so variable in time (e.g. pulses of metal effluent associated with storms) or space (e.g. bank erosion) that they are neither logistically nor economically feasible to monitor directly. Biological indicators also contribute to the development of qualitative water quality standards (e.g., surface water shall be free from floating oils) important for regulatory purposes. The three most common groups for monitoring programmes across aquatic ecosystem types are algae, macroinvertebrates and fish. Others, such as plankton and aquatic macrophytes, are commonly employed in specific ecosystem types such as lakes, reservoirs and wetlands.

Selecting relevant, diagnostic and meaningful indicators

The following section follows closely the recommendations of Jackson et al. (2000). An indicator is a sign or signal that relays a complex message, potentially from numerous sources using multiple parameters, in a simplified and useful manner. An ecological indicator is defined here as a measure, an index of measures or a model that characterizes an ecosystem or one of its critical components. An indicator may reflect biological, chemical, physical or hydromorphological attributes characterising the ecological condition. The primary uses of an indicator are to characterize current status and to track or predict significant change. With a foundation of diagnostic research, an ecological indicator may also be used to identify major ecosystem stress. Potentially, pressures and stressors can also be monitored and their evolution characterized by indicators.

There are several methods currently available for selecting indicators useful for estimating the ecological condition. They derive from expert opinion, assessment science, ecological epidemiology, national and international agreements, and a variety of other sources (see Noon, 1999; Anonymous, 1995; Cairns et al., 1993; Hunsaker and Carpenter, 1990 and Rapport et al., 1985). Regardless of the approach, useful indicators must provide information relevant to specific assessment questions, which are developed to focus monitoring data on environmental management issues. Numerous sources have developed criteria to evaluate environmental indicators. Jackson et al. (2000) assembled many of these into guidelines that provide for the evaluation of an indicator in four phases (originally suggested by Barber, 1994) (»Box 2.9). These phases describe an idealized progression for indicator development that flows from fundamental concepts to methodology, to examination of data from pilot or monitoring studies and lastly to consideration of how the indicator serves the programme objectives. The phased approach described allows interim reviews as well as comprehensive evaluations. Information used to conduct indicator evaluation is unrestricted and may include such things as journal articles, data sets, unpublished results and models, so long as they are technically and scientifically defensible.

There are hundreds of chemical, physical, biological and hydromorphological indicators that could be used to assess the ecological health of freshwaters and there is often considerable debate about the best methods (Norris and Hawkins, 2000; Hering et al., 2010). Monitoring has traditionally relied on water quality methods and structural measures of the biota, especially algae, invertebrates and fish, though trait-based indicators have also recently been proposed to provide greater diagnostic capability (e.g. Dolédec and Statzner, 2008). There is also growing recognition of the value of the ecosystem process measured in assessment programmes, as many goals of river management relate directly to the maintenance of natural ecological processes and these are often sensitive to causal factors that are known to affect river health (Bunn and Davies, 2000; Gessner and Chauvet, 2002; von Schiller et al., 2008).

Phase 1: Conceptual Relevance: The indicator must provide information that is relevant to societal concerns about ecological condition.

Phase 2: Feasibility of Implementation: Adapting an indicator for use in a large or long-term monitoring programme must be feasible and practical.

Phase 3: Response Variability: It is essential to understand the components of variability in indicator results to distinguish extrinsic/irrelevant factors from a true environmental signal.

Phase 4: Interpretation and Utility: A useful ecological indicator must produce results that are clearly understood and accepted by scientists, policy makers and the public (who are then ready to act upon information provided).
Source: Jackson et al. (2000)

Box 2.9: The four phases of indicator development

As a guiding principle, it is better to select a few indicators that are meaningful than to try and measure everything. Desirable freshwater ecosystem health indicators (adapted from Norris and Hawkins, 2000) are ones that:

- quantify threats and assets (drivers, pressures, stressors and responses);
- provide easily interpretable outputs;
- respond predictably to damage caused by humans;
- respond at appropriate timescales;
- are cost-effective to measure;
- relate to management goals; and
- are scientifically defendable.

An important goal is to select indicators that can help to diagnose the likely cause of observed degradation and guide management actions (Bunn et al., 2010). This is often difficult in freshwater systems where multiple stressors interact on different spatial and temporal scales to affect water quality, biodiversity and ecosystem processes (Townsend et al., 2003; Allan, 2004; Buck et al., 2004) depending on interrelated natural and man-made factors. Incorporating a range of indicators in monitoring programmes can help to identify whether impacts to stream and river ecosystems are the result of local factors (e.g. point source pollution), riparian or reach scale disturbances (e.g. stock access or riparian clearing), the effect of barriers downstream (e.g. Pringle, 1997) or the consequence of broader land use change in the upper catchment. Careful selection of indicators may help not only to diagnose the likely cause of degradation from a range of stressors, but also to identify the appropriate spatial scale for rehabilitation or protection. Without this diagnostic capability, monitoring programmes cannot be used with any confidence to guide management actions. »Box 2.10 illustrates the importance of large-scale indicators.

There is a major need to establish indicators that serve as surrogates of landscape-scale processes. Vegetated islands, for example, are common landscape features along river corridors and are at the same time among the first to disappear as a consequence of flow regulation and channelization. Islands integrate a variety of fluvial geomorphic and ecological processes (Ricaurte et al., 2012). Therefore, they serve as sensitive landscape-level indicators of the environmental conditions of river corridors. Furthermore, islands form critical nuclei for future conservation and restoration measurements within large rivers. They provide key resources, habitats and refugia, and, therefore, island-dominated river reaches exhibit a high resistance and resilience to natural and anthropogenic perturbation (Ward et al., 2000).

Box 2.10: Vegetated islands: Landscape-scale indicators of river corridors

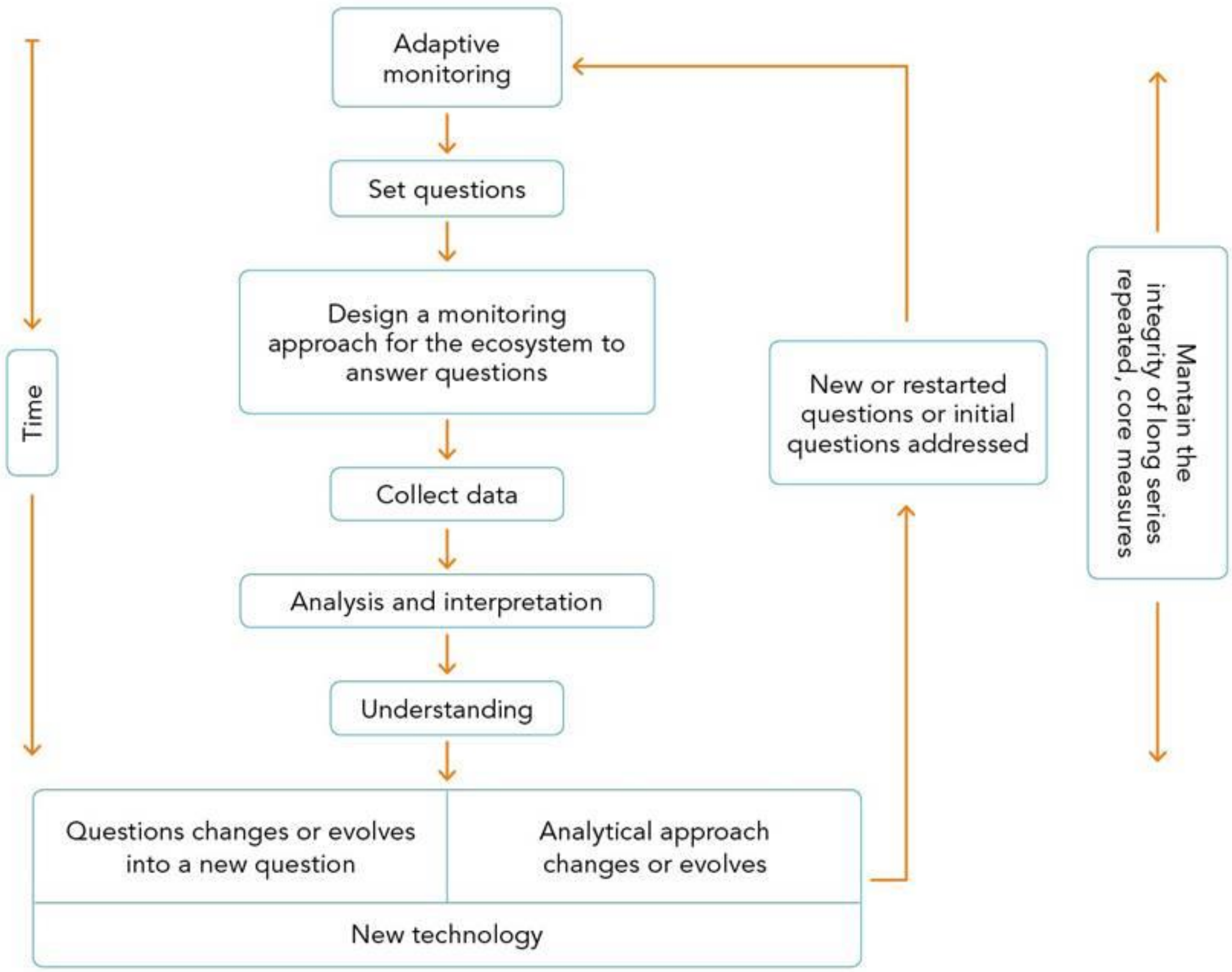
Establishing an ‘adaptive monitoring’ programme

Recently, there has been an increasing recognition of the need for monitoring programmes, specifically long-term monitoring programmes, to be adaptive in their set-up. Lindenmayer and Likens (2009, 2010) proposed an ‘adaptive monitoring’ paradigm, which fits well into the proposed adaptive management approach suggested in the Framework. As part of this paradigm, they identified what they believed to be the key features, or best practices, of successful and effective monitoring programmes. Programmes should:

- address well-defined and tractable questions that are specified before the commencement of a monitoring programme;
- be underpinned by rigorous statistical design;
- be based on a conceptual model of how an ecosystem might work or how the components of an ecosystem that are targeted for monitoring (e.g. a population) might function; and
- be driven by a human need to know about an ecosystem and thus are relevant to management.

A fundamental part of the adaptive monitoring paradigm is that the question setting, experimental design, data collection, analysis and interpretation are iterative steps (»Figure 2.10). A monitoring programme can then evolve and develop in response to new information or new questions. An important caveat here is that the adoption of new sampling or analytical methods can undermine the integrity of a long-term data record.

Figure 2.10: Adaptive monitoring. Adaptive monitoring provides a framework for incorporating new questions into a monitoring approach for long-term research while maintaining the integrity of the core measures. Initial key steps are the development of critical questions and a robust statistical design. Source: Adapted from Lindenmayer and Likens (2010a)



Evolving monitoring tools

In the last decade, research supporting the use of remote sensing approaches in monitoring programmes has greatly increased. Common water quality parameters measured using remote sensing approaches include temperature, chlorophyll concentration, turbidity, and salinity (e.g. Shafique et al., 2003; Focardi et al., 2006; Somvanshi et al., 2012; Akbar and Hassan, 2013; Papoutsa and Hadijmitsis, 2013). Remotely sensed data is also being used for the assessment of many physical habitat variables such as vegetation cover, land cover, geology, bank angle and many others (e.g. Gilvear et al., 2004; US EPA, 2006a; Williams et al., 2013). Compared with water quality parameters, these physical habitat measures have not been broadly incorporated into monitoring programmes because of a lack of scientific consensus on which methods to use and which river features to monitor. This is partially because many fundamental questions relating hydrological, geomorphological and biological characteristics remain unanswered, and because river habitats are monitored to cover a wide variety of objectives (Walczykowski, 2013). However, the cost-effectiveness and the opportunity to address questions on increasingly large scales will most certainly lead to their more complete development.

Another area of evolving monitoring tools is next-generation DNA sequencing techniques that promise to improve and advance environmental monitoring and biological assessment (Hajibabaei, 2011; Pilgrim et al., 2011; Sweeney et al., 2011). Currently, when biological indicator data is collected, the morphological identification of organisms may often require significant time and resources. Molecular methods such as DNA barcoding can help with this process by working to not only reduce the time it takes to identify organisms, but also to help with the identification of difficult taxonomic groups (e.g. algae, bacteria). These approaches also represent viable options for processing biological samples where local taxonomic expertise is not available. DNA barcoding uses short genetic markers in an organism's DNA to identify its species. These techniques can be used to help determine the condition of specific aquatic ecosystems, detect whether a site is improving or degrading and may even help scientists detect the presence of pollutants in streams.

Genetic sequencing methods also allow the metagenome of multiple (microbial) species to be characterized in situ. Both taxonomic and functional biodiversity of microbial assemblages can be measured simultaneously, without necessarily having to know which species are present (Friberg et al., 2011). As these and other emerging techniques are introduced into monitoring programmes, it will be essential to initially test them in parallel with traditional metrics to be able to assess their relative merits and to provide a means for calibration across existing and emerging methods (Friberg et al., 2011).

Key requirements of monitoring and study design elements

This section follows closely the recommendations formulated by the US EPA for biological assessment processes (US EPA, 2006b). However, they illustrate the pre-requisites for monitoring in general. Study design is the foundation of any monitoring programme. Design elements include study design objectives, temporal and spatial coverage, classification, RCs and criteria (US EPA, 2006b). Design questions may be driven by regulatory requirements, programme goals and research questions. In any case, it is strongly recommended that analytical or statistical specialists cooperate with field experts and programme managers during this phase. Too often, assessment programmes are created without a clear sense of how the data is going to be used, only to discover that the design chosen was inappropriate to provide the answers or data quality needed to meet assessment programme objectives (Lindenmayer and Likens, 2009, 2010a). However, it is also possible to create Data Quality Objectives (DQOs) that are unattainable or technically too difficult to implement.

Design questions are usually derived from programmatic needs. Ecosystem assessment data is used in a variety of programmes (e.g. reporting, source water assessments, permitting[24]). Developing a monitoring programme that meets multiple needs requires an understanding of the information needed by each programme and, thus cooperation between programme personnel. Although this involves effort, cooperation at this point can help maximize utility of the data for multiple purposes and minimize the collection of redundant data by multiple programmes.

While developing the design questions, it is also critical to develop DQOs to determine the quantity and quality of data needed (US EPA, 2000b). DQOs are quantitative and qualitative statements that clarify objectives, define appropriate data and specify tolerable levels of decision error. Each DQO has to be described to define the design elements needed to satisfy the assessment precision and accuracy required by the programme. For example, programmes needing only to separate extremely disturbed from minimally disturbed sites will require less precision than programmes designed to detect small departures in ecosystem condition (»Figure 2.11). Some ecosystem types

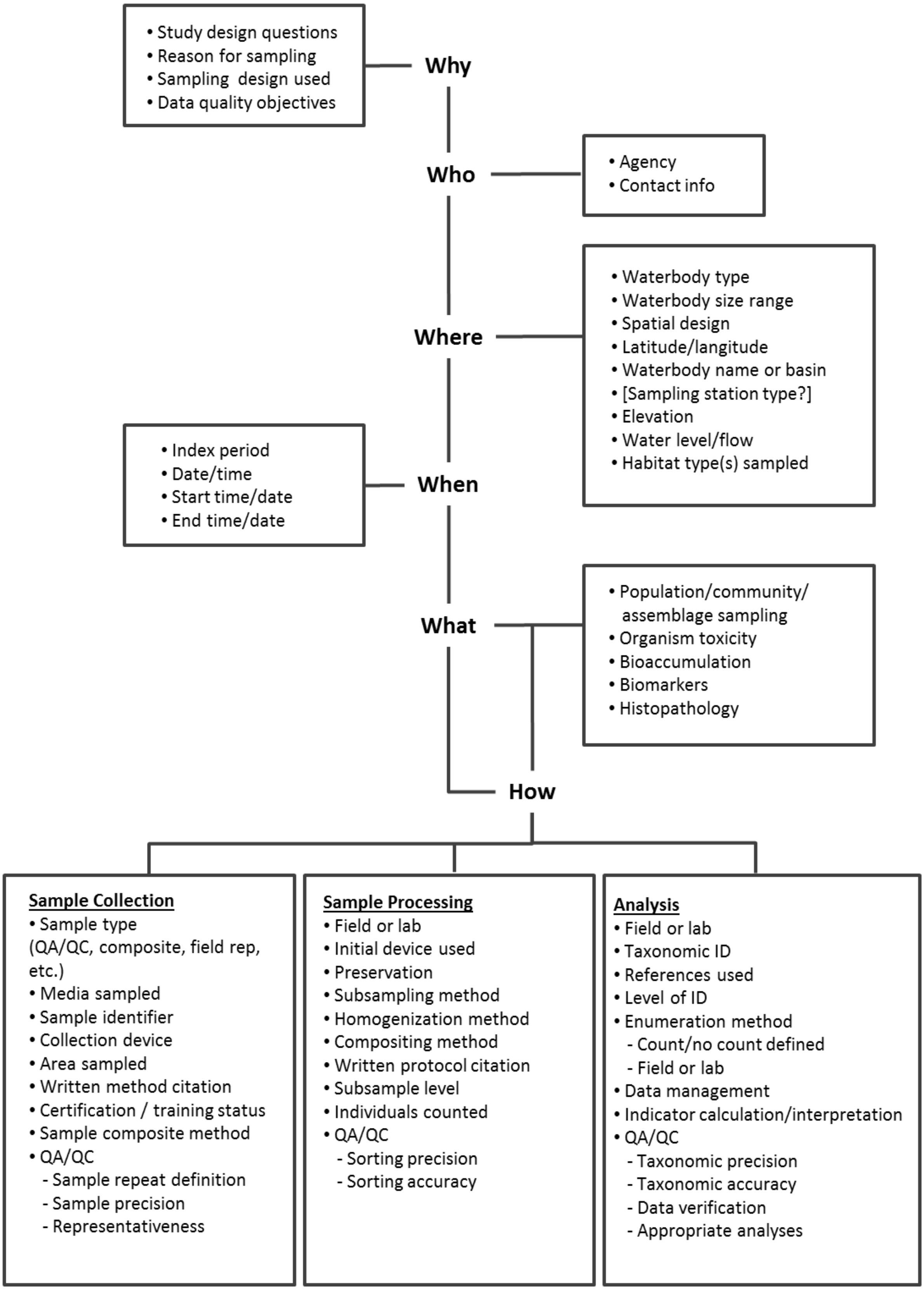
[24] It has to be kept in mind though that some developing countries may not dispose of a permitting law.

may require greater precision (e.g. larger rivers) since the range of existing biological conditions, in many cases, is already restricted (US EPA, 2006b).

It is also important to include performance evaluation as a part of the monitoring programmes from the beginning. This includes being able to document and report the quality of each step from data collection to site assessment. Performance elements (precision, accuracy/bias, representativeness, completeness and sensitivity) must be included in the study design and incorporated into assessment programme costs. Measurement Quality Objectives (MQOs) document method performance, as well as programme technical staff, and are measurement goals needed to meet the programmatic DQOs. In general, MQOs do not specify the methods, but provide criteria for evaluating acceptability of data produced by a protocol or programme (US EPA, 2006b; Stribling, 2011).

Precision, calculated on final assessments, can be used to identify errors and to determine the repeatability of ecosystem assessments. For example, assessment precision is generally evaluated using repeat sampling for some sites by the same team (to evaluate intra-team precision) or by different teams (to evaluate inter-team precision) (Barbour et al., 2006). Precision also affects the ability of a method to detect an effect.

Figure 2.11: Data elements for biological assessment programmes.
Source: Modified from US EPA (2006b) and NWQMC (2006)



Setting Benchmarks for Ecosystem Health

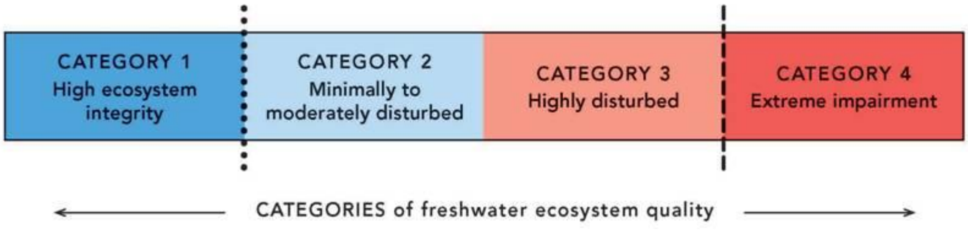
The most basic scientific and policy task of water management programmes is to agree on what distinguishes "good" (acceptable) from "bad" (unacceptable) conditions. In general, the thresholds demarcating acceptable from unacceptable chemical, physical and biological conditions are referred to as benchmarks. Other terms used in specific contexts, especially with regulatory implications, include guidelines, criteria and standards, and their meaning varies between programmes and governmental organizations. From an ecosystem perspective, it is important to set high integrity (i.e. 'naturalness') as an aspirational goal. This reflects a societal objective of maintaining high ecological integrity (i.e. notions of 'intactness' or 'naturalness'), with the assumption that biodiversity and key ecological processes are protected and maintained. Given the objectives of the Framework for Freshwater Ecosystem Management (the Framework), there is unlikely to be any disagreement that where possible the lowest benchmark for ecosystem protection should still correspond with a quality status that supports a balanced, integrated and adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of a region (Frey, 1975).

In some regions, it may be possible to set upper targets relative to a certain 'reference' condition, a condition undisturbed by human activities that can serve as an anchor point for comparison, such as applied in the Australian River Assessment System (AUSRIVAS) (Schofield and Davies, 1996) and the River Invertebrate Prediction and Classification System (RIVPACS) (Wright, 1995). This of course assumes that there are places in the landscape that have been relatively free from intensive human activity or there is historical information available to reflect conditions prior to human influence. For indicators of land use stressors, we can use modelled information to predict the RC. For some water quality indicators, such as pesticides or other novel compounds, we can safely assume the reference value was 'zero'. Similarly, for biological indicators such as the percentage of alien species, we can also assume that a reference value was 'zero'.

Such RCs do not necessarily have to constitute an attainable target condition, but they provide transparency about how far an ecosystem has departed from its natural state. This information helps the manager and the public to better understand what is at stake in their decision making relative to remaining high quality conditions and setting realistic restoration targets in areas of widespread human disturbance. In contrast, setting a low target would concede that it is acceptable if ecosystem health is simply maintained above some minimum value and this would in consequence increase the risk of further biodiversity loss and diminished ES. Setting a high target also provides incentives to identify and protect systems with high ecological integrity, recognizing that the cost of prevention of degradation is likely to be substantially less than the cost of restoration. »Figure 2.12 provides an example of how the range in acceptable conditions for a given ecosystem might be characterized. Setting upper (Category 1) and lower ('dead-zone', i.e. extreme impairment, Category 4) benchmarks provide important 'anchor points' for condition indicators. In this scheme, RC (if known) would be "extreme left" at the upper limit of Category 1. Given the objectives of the Framework, we would assume that the lowest benchmark would be the upper threshold of Category 4 – i.e. at a minimum, it is essential that the ecosystem at least supports some higher multi-cellular life forms. In the case of water quality parameters, these would be values known to be acutely toxic to algae, invertebrates and/or fish. For biological indicators, such as species richness or diversity, these would be values above zero.

However, thresholds of concern between these two extremes that would trigger management actions if reached (illustrated between the dashed and dotted vertical lines in »Figure 2.12) also need to be established: An upper benchmark that warns of departure from high ecosystem integrity to a modified state and a lower benchmark that warns of approaching an unacceptable state. These benchmark levels are also sometimes called "thresholds of concern" in the respective scientific literature.

Figure 2.12: Categories of freshwater ecosystem quality.
The dotted and dashed bars indicate the two benchmarks discussed in the text.



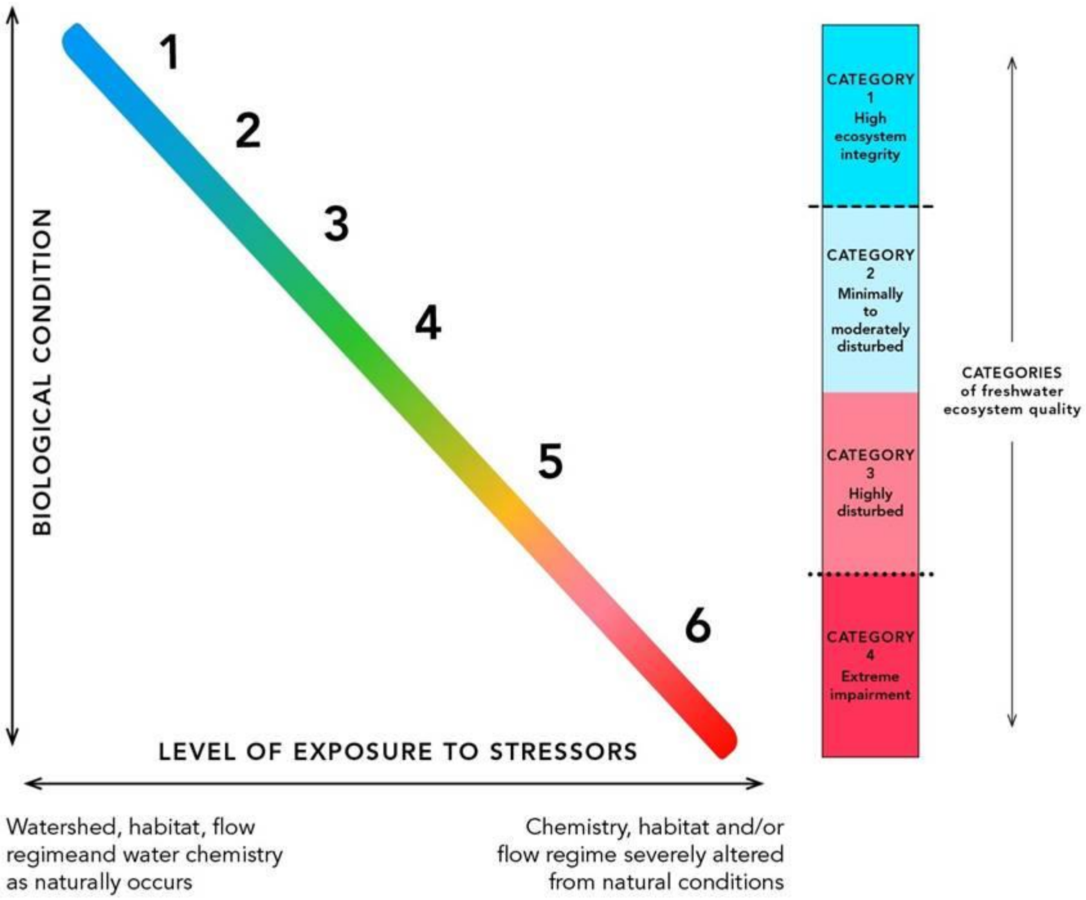
Setting appropriate upper target values is not always straightforward given the often unalterable modifications that have been made in the landscape in the past. In situations of human dominated landscapes (e.g. channelized streams and rivers, widespread agricultural and urban development, etc.), which are the product of interactions between societies and ecosystems over many thousands of years, setting an upper target of undisturbed or minimally disturbed 'reference' condition may not be feasible. Accommodations must be made to account for these changes while providing for setting best attainable ecological conditions in current circumstances and best practices for remediation and management.

Stoddard et al. (2006) provide an extensive discussion of the RC concept presenting various approaches; two are described below. One approach is to set upper targets relative to the 'Least Disturbed Condition' (LDC). However, for some freshwater types (e.g. many lowland rivers), this LDC may already be highly disturbed by an activity. A more objectives-based approach is to set targets for 'Best Attainable Condition' (BAC), which has been defined as the condition that could be achieved by implementing best management practices (Stoddard et al., 2006). BAC is not to be confused with a management objective where tradeoffs and compromise are made between ecosystem health and the costs of management intervention to protect or restore. The BAC rather acknowledges the presence of humans in the landscape and considers what is technically possible, if, for example, point source pollution is eliminated, diffuse pollution reduced by best practices in urban and rural catchment management or degraded habitats (fringing vegetation, in-stream, etc.) are rehabilitated. In most settings, such kinds of action should enable a high level of ecological remediation to be achieved, which corresponds to the BAC.

The scientific methods and inferences for developing benchmarks differ depending on whether the assessment endpoint is for an exposure (e.g. chemical or physical water quality criteria) or for a desired biological condition such as the diversity and abundance of aquatic life (biocriteria). The specific methods used for each vary globally. »Annex 3 provides an example from the United States and includes a discussion of the useful scientific elements of water quality benchmarks (WQB); the importance of setting baselines for comparison during development and implementation; and some of the more common methods for deriving laboratory-, field-, and model-based benchmarks. Additionally, the value and use of descriptive models for interpreting results across differing programs and for communicating effectively with the public and policy makers about the current status of aquatic resources and their potential for recovery to a more desired condition are discussed. The Biological Condition Gradient (BCG) (see »Annex

3 for more detail) is a very informative classification of different quality states of freshwater ecosystems. »Figure 2.13 compares the BCG with the proposed freshwater ecosystem quality categories.

Figure 2.13: The Biological Condition Gradient.
Source: Modified from Davies and Jackson (2006) and,US EPA (2011b).



Synthesis reporting and communication of results

The results of monitoring and assessments of conditions relative to goals have to be communicated to different target audiences, but audiences' requirements and their further use of the information may be quite different. A common requirement when communicating with any audience is the necessity to aggregate both qualitative and quantitative scientific results into coherent messages, graphics or other forms of metrics and ranking. Gradient concepts are integral to the ecological status objectives of, for example, the EU WFD. The EU WFD and the US CWA aim at the biological and physico-chemical protection or restoration of water bodies. For example, the EU WFD's goal of 'good ecological status' for all waters, as stipulated by the EU WFD, is comparable to the goal for aquatic life protection as defined in the US CWA. However, the latter defines its goals broadly and authorizes flexibility to States to implement and manage its water quality. The US CWA does authorize the United States to develop and manage their water quality programme with oversight and review by the US EPA though for consistency with the US CWA goals and objectives. The specific goals of the EU WFD are defined using prescribed parameters indicative of all biological, hydromorphological and general physico-chemical quality elements that have to be monitored.

As introduced in »Section 2.7, the BCG method offers a process to develop a shared understanding and "guiding image" (Willby, 2011) of restoration goals in relation to current ecological status. The model helps non-scientists and resource managers to interpret and communicate the relative implications of complex ecological data. The comparison of EU WFD quality classes and BCG levels reveals a high degree of similarity between both frameworks (Reitberger et al., 2010). Both these frameworks, and others like them, provide help to resource managers with goal-setting, the design of monitoring and assessment programmes, strategic planning, and the setting of protective criteria against stressors. »Annex 3 of this document provides biological and other ecological attributes used to characterize the BCG, tiers of the BCG as well as ecological attributes by condition tiers matrix.

An essential element of monitoring and assessment is to inform those responsible for policy and management so they can respond and address emerging issues. It may also be desirable to communicate findings more broadly to engage interest and support from the public for management intervention. Reporting involves the effective communication of key findings of monitoring and assessment, and is an essential process in closing the adaptive management loop.

Report cards are an effective communication tool and aim to increase accountability by reducing "information asymmetry" between society and government (Coe, 2003). Report cards have long been used in education to communicate student progress to parents, but since the 1980s have been increasingly used to report on the state of the environment and economy, and more recently, the human health system. The benefits of employing environmental report cards include raising environmental awareness, engaging citizen scientists and catalysing management decision making. They have been used with great success to communicate the results of river health monitoring programmes, for example in Australia, China, Papua New Guinea and the US (see Dennison et al., 2007 and http://www.ehmp.org; http://www.watercentre.org/research/rhef/project-resources/report-cards/reportcards for examples).

Governance and legal issues

Adaptive governance and management frame

As a response to environmental degradation, regulatory frameworks increasingly address environmental issues. Most countries have provisions in water-related legislation addressing the state of the environment. However, encompassing regulatory frameworks are a necessary, but not sufficient, condition for effective policy implementation (Pahl-Wostl et al., 2012). Often, countries face serious implementation gaps. These may derive from a lack of human and financial resources and technical skills, from ineffective administrative procedures or asymmetric power structures. Guidance as provided in this report can help to support more effective policy implementation.

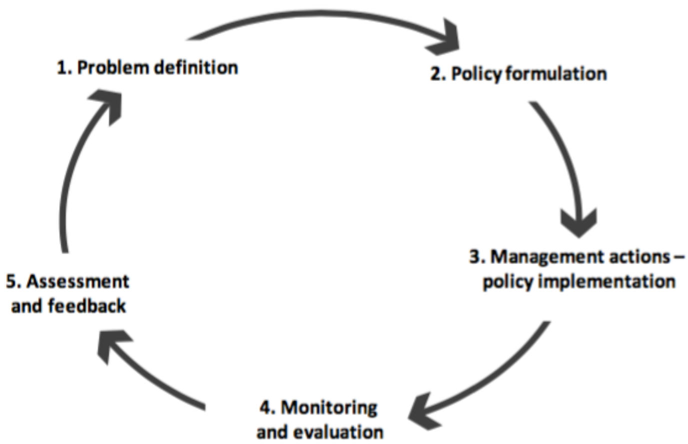
Respect of good governance principles during policy development and implementation is essential for effective policy implementation. Good governance is participatory, accountable, transparent, responsive, consensus oriented, effective and efficient, equitable and inclusive, and follows the rule of law (UNESCAP, 2016).
Uncertainties are associated with implementation. Ecosystems are complex and may respond to management interventions in unexpected ways. Climate change, economic and population development lead to an alteration of the context within which water management has to perform. Adaptive management is essential to deal with such uncertainties and to support a systematic and structured process of learning.

Adaptive management can be visualized as an iterative cycle of policy development and implementation as represented in »Figure 2.14. Monitoring and evaluation of management actions and policy implementation are essential for learning from experience. The adaptiveness does not derive only from the cyclic iterations, but also from the very nature of the different phases (Pahl-Wostl et al., 2007). Uncertainties need to be taken into account in each step.

The interest in adaptive management has mushroomed with the increasing awareness of the need for climate change adaptation. However, the importance of adaptive management does not derive only from climate change impacts. Managing complex systems as social-ecological systems in river basins typically requires adaptive and flexible management approaches to be able to respond to complex dynamics and unexpected developments.

Pahl-Wostl et al. (2007) summarized the key requirements for realizing adaptive management. New information must be available and collected, for example performance indicators and indicators of change that may lead to desirable or undesirable effects. They are to be monitored over appropriate timescales that are definitely longer than those mandated by short-term political objectives. The actors in the management system must also be able to process this information and draw meaningful conclusions from it. This can be achieved if a learning cycle and negotiation process unite the actors in all phases of assessment, policy implementation and monitoring. As actors pursue different and changing political interests, transparency and leadership are of major importance to make such multiparty negotiation processes work. Change must be possible in ways that are open and understandable to all actors. Managers must be able to implement change based on new information, processed in a transparent manner. Transparency implies that procedural rules exist and that there is clear communication as to who decides how and when to change management practices and what evidence was used to make this decision. To do this, it is necessary to strike a balance between continuity and flexibility because some management strategies may take one or more decades to be implemented and tested.
European examples are used to illustrate how far conducive political frameworks help to establish governance schemes. These examples, while certainly showing spectacular success, also reveal the need for an underlying consensus of stakeholders, the long timescales involved and the considerable and continuous financial engagement as necessary pre-requisites of the sustainable protection and restoration of freshwater ecosystems (OECD, 2011).

Figure 2.14: Iterative cycle of policy development and implementation in adaptive management.
Source: Modified from Pahl-Wostl et al. (2007).



Evolution of water governance – the example of the European Union

Water legislation in the EU evolved from focusing on problems in isolation towards more integrative and participatory approaches. This change coincided with a shift from

i. an emphasis on “end-of-pipe solutions” and the prescription of both goals and the means of achieving them towards
ii. addressing causes and leaving more options to define operational targets and the means of achieving them during policy implementation.

This evolution is outlined in the following paragraphs for the development of water policy in European countries. Since the 1970s, this development has been guided by policy at European level[25].

Aubin and Varone (2004) gave an account of the evolution of European Water Policy over the past decades which is typical of the development of environmental policies in Western European countries in general. The phase of the first generation of directives (1970s to early 1980s) mainly set quality standards for anthropocentric water uses. Examples are the European Drinking Water Directive (Council Directive 98/83/EC) and the Bathing Water Directive (Council Directive 2006/7/EC). The phase of the second generation of water directives (until about 1995) adopted a command and control approach that focused on the limitations of emissions of certain categories of pollutants. A central problem was the eutrophication of inland and coastal waters caused by nitrates and phosphates. The European Urban Wastewater Directive 91/271/EEC of 1991[26] targeted domestic discharges. It prescribed the implementation of WWTPs and in some regions technical specifications (denitrification). The diffuse pollution by nitrates from agricultural sources was targeted by the European Nitrates Directive Directive 91/676/EEC of 1991[27]. Member states were entitled to set priorities in particularly vulnerable regions and limit the spread of manure and chemical fertilizers. The directive emphasized developing codes of good practice and their voluntary adoption by farmers. However, the effectiveness of measures was not as expected. Decentralized and voluntary measures did not give satisfactory results. Agricultural policy reforms were also not supportive in encouraging farmers to adopt the recommended practices[28].

With the EU WFD, European water policy finally entered into an entirely new area of water legislation. The EU WFD, which entered into force in the year 2000, has the following key aims (EC, 2000a):

- Expanding the scope of water protection to all waters, surface waters and groundwater,
- Achieving ‘good status’ for all waters by a set deadline,
- Water management is based on river basins,
- ‘Combined approach’ of Emission Limit Values (ELV) and quality standards,
- Obliges to have pricing policies,
- Getting the citizens involved more closely,
- Streamlining legislation.

The EU WFD is quite ambitious and requires considerable resources and skills for its implementation. However, its general principles are based on several decades of learning and experience with a variety of approaches for dealing with water quality problems. Furthermore, it is designed to give Member States the possibilities to tailor the implementation process to their needs taking into account the specific conditions in the different countries. Table 2.2 summarizes this evolutionary process.

While the example of the EU may not be the model for all national or international

[25] Please note that many EU Member States have legislation previous to 1970 which also guides their policy currently. Their legislation could even go beyond that of the EU.
[26] http://ec.europa.eu/environment/water/water-urbanwaste/index_en.html
[27] http://ec.europa.eu/environment/water/water-nitrates/index_en.html
[28] Furthermore could have Rural Development Plans been envisaged by farmers on a voluntary basis

cases, it does indicate the gradual move from a utilitarian towards an ecocentric focus. Furthermore, it emphasizes potential time scales (decades), the pitfalls of top-down approaches in objective and policy formulations as well as the importance of enforcement mechanisms in implementation. It is also obvious that directives and

guidelines are not to be developed and issued for their own sake, but should support the achievement of societal consensus-based objectives, such as securing ecosystem health while meeting the water demands of the different sectors of users.

Table 2.2: Policy design of the successful generation of EU water governance and directives. Source: Modified after Aubin and Varone (2004), page 64

Issues to be addressed	1st generation	2nd generation	EU WFD
Logic of intervention	Immission Limit Values, Water quality standards	Emission Limit Values (ELV), Emission control	Combined approach
Water bodies	Bathing water, fish water, (etc.), Groundwater, Human-centred logic	Water resource	Eco-centred logic
Uses	Drinking water, Transport and absorption, Navigation	Living environment, Drinking water, Production, Transport and absorption	Living environment, Drinking water, Production, Transport and absorption, Navigation and support, Recreation
Objectives	Preserve human health, Protect particular water uses, Harmonization of national legislation, Limit emissions of substances that cannot be diluted	Preserve human health, Fight against euthrophication, Put the most dangerous pesticides out of the market, Reduce industrial discharges, Preserve biodiversity and habitats	Good status for all waters, Management on a river basin scale, as of 2015
Instrument mix	Prescription and information (Minimum quality requirement from which ELV are set, Harmonization of controls, Prohibition of listed dangerous substances, National programmes and reports)	Prescription information and self-regulation (Prohibition/authorization of discharges, Timetable for wastewater treatment sensitive areas, List of substances, Harmonization of labelling, Codes of good practice, Action programme and monitoring, Consumer information)	All instruments, Including economic incentives in the management plan inventory, Programme of measures, Integration by full cost pricing
Actors of implementation	Member state and the EC for some aspects (Adoption of ‘daughter’ directives, revision of the emission standards)	Member states and the EC/EU (Revise parameters)	Multi-level (EU Member States and basin authorities) and multi-sector (environmental, economic and public works)
Target groups	Industry, Drinking water producers, Housing, Nuclear industry	Farmers, Pesticide producers and users, Industries (nutrients and polluting substances), Public authorities in charge of sewage, Drinking water producers	Industry, Households, Farmers, Navigation (inland and sea)

Final beneficiaries	Drinking water consumers, End users (swimmers, fishers, etc.), Industries from the most advanced Member States, Fauna and flora	Drinking water consumers, Water industry, Drinking water producers, Other water users, Fauna and flora	All humans users, Fauna and flora

Transboundary contexts

There is a large number (276) of shared major river basins accommodating about 40% of the world's population. Well over 300 transboundary aquifer systems provide ES for about 2 billion people[29] (UNESCO, 2013). Therefore, the consideration of transboundary issues and cooperation is a primordial pre-requisite of successful water management in general and of the protection and rehabilitation of freshwater ecosystems in particular. The scale of water governance, whether it is local, national, or transboundary, is an important factor that can hinder the development of water resources management strategies. At national level, interests and power struggles can thwart appropriate cooperation in the decision-making process related to transboundary water governance due to diverse interests, uneven distribution of control and/or decision making (Dore et al., 2012). Water quality problems in international river basins cannot be solved at national level only. In particular, downstream riparian countries are affected by the activities of upstream countries.

Globally, more than 260 rivers, covering 45% of the land surface (excluding Antarctica), are transboundary (Wolf et al., 1999). These rivers drain areas of up to 18 countries (Danube basin), thereby integrating across a vast range in ecological, political, social and economic conditions. Only a fraction of these transboundary rivers and their water resources are effectively managed by international organizations. Recent analyses suggest that high water variability within transboundary basins fosters cooperation among the countries. However, climate change may further increase variability and therefore may have destabilizing effects on international river management. Variability is particularly high in Africa, as well as in major parts of Asia, putting transboundary rivers in these regions at future risk of experiencing conflict and reduced cooperation (Dinar et al., 2015).
In Southeast Asia, for example, the six major transboundary river basins that drain South China include the Red River, Mekong, Salween, Irrawaddy, Ganges-Brahmaputra-Meghna and Indus. A recent analysis by Kattelus et al. (2015) demonstrates that, while China has a fairly low level of vulnerability in these basins, the downstream consequences of upstream alterations may be immense, which demands a high level of responsibility of the upstream countries. Indeed, these basins will face one of the most serious water infrastructure developments within the coming decades, which demands for transnational cooperation and agreements.

Box 2.11: Transboundary river management

In many transboundary river basins, formal cooperation among countries is not yet advanced. UNEP-DHI (2016) concluded that effective formal transboundary treaties that recognize key principles of international law are more frequent in Europe and North America than in Asia, Africa or South America. However, there is no doubt that in many river basins of the world the achievement of water quality targets hinges on transboundary cooperation. Europe provides an example of particularly successful cooperation across national borders. However, many river basins in the world are located in regions where transboundary cooperation cannot build on such supra-national frameworks. In such cases, cooperation on water problems can be pioneering and also facilitate cooperation in other fields of common concern rather than triggering

[29] http://www.transboundarywaters.orst.edu/about/wolf.html; http://programme.worldwaterweek.org/event/how-the-two-3637

water conflicts. A first step is to overcome the absence or failure of formal negotiations at national levels. This may be cooperation on information among water management

authorities in different riparian countries. Such cooperation can build trust which is an impetus for further steps to develop and implement transboundary agreements (Raadgever and Mostert, 2005).

Collaboration on water pollution issues was a key driver for international collaboration in the Rhine basin, which is often cited as a model for successful trans-boundary water management (UNESCO, 2013). Requirements for successful transboundary collaboration are binding international agreements, the implementation of an intergovernmental coordination body, transboundary basin organization with sufficient resources and a mandate for the coordination of measures and the monitoring of progress. Since the 1997 UN Water Convention on Non-navigational use of Transboundary Waters achieved obligatory status in 2014[30] upon ratification by the necessary number of Member States, the appropriate paragraphs on the "no harm" or obligatory information principles provide the adopted international framework for riparian states to synchronize their activities. For further information, see »Box 2.11.

Examples for treaties and cooperation on international basins

With the signing of the agreement on the International Commission for the Protection of the Rhine against Pollution (Berne Convention) in 1963, the States bordering the Rhine created a basis under international law for their cooperation. Gradually, this Convention received a broader scope and several decades of negotiation and increasing cooperation among riparian countries finally led to the encompassing and pioneering Convention on the Protection of the Rhine.

Convention on the Protection of the Rhine
The Convention on the Protection of the Rhine (signed in 1999, entered into force in 2003). The aims of the Convention are sustainable development of the Rhine ecosystem through:

- maintaining and improving the quality of the Rhine's waters, and its natural function;
- protecting species diversity;
- reducing contamination;
- conserving and improving natural habitats for wild fauna and flora;
- ensuring environmentally sound management of water resources;
- taking ecological requirements into account when developing the waterway;
- production of drinking water;
- improvement of sediment quality;
- flood protection;
- coordination with measures to protect the North Sea. (recipient sea of the River Rhine).

The Convention defines a constitutional basis for the implementation and operation of the International Commission for the Protection of the Rhine. Regarding the improvement of the state of the riverine ecosystem, the ambitious coordinated programme "Rhine 2020" emphasizes the achievement of ecological improvements. These encompass the restoration of floodplains, increase of habitat connectivity and improvement of ecological continuity. The shift in emphasis from water quality to ecological characteristics reflects the insight that healthy ecosystems need more than improved physico-chemical properties. Furthermore, this shift has been triggered by the EU WFD, which entered into force in the year 2000 (see also »Section 2.9.2). The EU WFD´s goal is to achieve a good ecological state for all European freshwater bodies.

Danube River Protection Convention

[30] http://www.transboundarywaters.orst.edu/about/wolf.html; http://programme.worldwaterweek.org/event/how-the-two-3637

The Danube River Protection Convention (DRPC)[31] (signed in 1994, entered into force in 1998) forms the overall legal instrument for co-operation on transboundary water management in the Danube River Basin.

The main objective of the DRPC is to ensure that surface waters and groundwater within the Danube River Basin are managed and used sustainably and equitably.

This involves:

- the conservation, improvement and rational use of surface waters and groundwater
- preventive measures to control hazards originating from accidents involving floods, ice or hazardous substances
- measures to reduce the pollution loads entering the Black Sea from sources in the Danube River Basin.

The signatories to the DRPC have agreed to cooperate on fundamental water management issues by taking "all appropriate legal, administrative and technical measures to at least maintain and where possible improve the current water quality and environmental conditions of the Danube river and of the waters in its catchment area, and to prevent and reduce as far as possible adverse impacts and changes occurring or likely to be caused."

The TransNational Monitoring Network (TNMN) was established to support the implementation of the DRPC in the field of monitoring and assessment. The TNMN was formally launched by the International Commission for the Protection of the Danube River in 1996, although the history of international monitoring of the Danube River is much longer. The main objective of the TNMN is to provide a structured and well-balanced overall view of pollution and long-term trends in water quality and pollution loads in the major rivers in the Danube River Basin.
The TNMN utilizes nationally assessed monitoring data and is based on the provisions of the DRPC, which requires:

- Harmonizing monitoring and assessment methods, particularly concerning water quality in rivers,
- Developing coordinated or joint monitoring systems applying stationary or mobile measurement
- devices, and shared communications and data processing facilities,
- Elaborating and implementing joint programmes for monitoring riverine conditions in the Danube catchment area, including flow rates, water quality, sediments and riverine ecosystems, as a basis for the assessment of transboundary impacts.

In 2006, the TNMN was revised to ensure full compliance with the provisions of the EU Water Framework Directive (WFD). The TNMN monitoring network is based on national surface water monitoring networks and includes 79 monitoring locations with up to three sampling points across the Danube and its main tributaries. The minimum sampling frequency is 12 times per year for chemical parameters in the water and twice a year for biological parameters. An interim water quality classification scheme has been specially developed to evaluate the data collected by the TNMN. The assessment of loads in the Danube contributes greatly to estimates of the influx of polluting substances to the Black Sea and provides vital information to support policy development. A special load assessment programme was started in 2000, with pollution loads calculated for BOD5, inorganic nitrogen, ortho-phosphate-phosphorus, dissolved phosphorus, total phosphorus, suspended solids, and - on a discretionary basis – chlorides.
Source: https://www.icpdr.org/main/activities-projects/tnmn-transnational-monitoring-network

Box 2.12: Transnational Monitoring Network in the Danube Basin

The International Commission for the Protection of the Danube River (ICPDR) acts as a platform for its contracting parties to coordinate responses to various environmental threats. The Danube River Management Plan has been developed in line with the implementation of the EU WFD. It provides the roadmap for action and includes a Joint Programme of Measures. There is no doubt that the EU with its legislation is substantially facilitating and

[31] Source – webpage of the International Commission for the Protection of the Danube River. http://www.icpdr.org/main/icpdr

enhancing the implementation of transboundary agreements and efforts within its jurisdiction. The majority of both the Rhine and the Danube basins are located within the EU. The adoption of the EU WFD and other environmental legislation was among the pre-requisites new EU Member States had to implement before joining the Union. The EU WFD prescribes achievements and monitors compliance on a national scale. However, the harmonization of targets and monitoring procedures brought about by European legislation facilitates transboundary cooperation. For further information, see »Box 2.12.

Mekong River Commission (MRC)
In the Mekong River Basin (MRB), issues of governance are particularly challenging due to the fact that the Mekong River crosses six countries, but only the four downstream countries are members. The MRC was established in 1995 by the Mekong Agreement, succeeding the Mekong Committee and the Interim Mekong Committee, which were active since 1957 and 1978 respectively. The Mekong River Commission (MRC) is an inter-governmental facilitating and advisory body governed by water and environment ministers of Cambodia, Lao PDR, Thailand and Vietnam to jointly manage the shared water resources of the Mekong River[32] and ensure that Mekong water is developed in the most efficient manner that mutually benefits all Member States. In the 1995 document, participating countries agreed that any actions taken in the MRB must ensure that minimum monthly flows remain acceptable, wet season reverse flow into Tonle Sap Lake must be maintained and floods must not be exacerbated. Cooperation is a norm, but, when occasional conflicts arise (such as in the case of dam development), the MRC provides a framework and platform to cooperate and discuss issues of basin-wide relevance. Practical achievements include a set of processes and strategies that allows the countries to discuss technical aspects of the development and management of the basin's resources (UNESCO, 2013) such as the Procedures for Water Quality in 2011, to align the implementation of activities to monitor water quality and ecological health[33]. Further MRC guidelines exist on the Implementation of the Procedures for Water Use Monitoring[34]. The example of the Mekong illustrates the challenges but also successes of transboundary cooperation in a setting where riparian countries differ considerably regarding their political systems and state institutional and economic development.

Bilateral Slovak – Hungarian transboundary surface water monitoring programme
Since 1976, the Slovak Republic and Hungary have cooperated on the monitoring of their transboundary rivers. From 1989, the broad complex surface water monitoring has been established between countries. The monitoring programme consists of a sampling site list with coordinates, a list of more than 70 parameters (physico-chemical, chemical, hydobiological, microbiological and radiological parameters), frequencies, methods of sampling and analysis. Results are evaluated in a harmonized way each year and presented in the form of a Yearly Common Final Report (annual reports only available in Hungarian and Slovakian; further reading: Kovács et al., 1998).

UN conventions
UNECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes[35]

The United Nations Economic Commission for Europe (UNECE) has to tackle a wide range of water quantity and water quality problems: High water stress and overexploitation of water resources, increasing droughts and floods, contaminated water resulting in water-related diseases, etc. Attempts at solving these complex problems in Europe are further complicated by the transboundary nature of water resources. More than 150 major rivers and 50 large

[32] Source – webpage of the Mekong River Commission (MRC). http://www.mrcmekong.org/about-the-mrc
[33] http://www.mrcmekong.org/assets/Publications/policies/Procedures-for-Water-Quality-council-approved260111.pdf
[34] http://www.mrcmekong.org/assets/Publications/policies/Tech-Guidelines-PWUM.pdf
[35] Source of explanations: http://www.unece.org/env/water/text/text.html

lakes in the UNECE region run along or straddle the border between two or more countries. Fortunately, UNECE Member States are aware of the need for cooperation if they are to ensure that transboundary waters are used reasonably and equitably. They know that they share the same water resources and rely on each other in order to apply effective solutions. This positive attitude to the problem has been triggered, in no small measure, by the UNECE Convention on the Protection and Use of Transboundary Watercourses and International Lakes (this entered into force in 1996), which 36 UNECE countries and the EU have already ratified.

The Convention takes a holistic approach based on the understanding that water resources play an integral part in ecosystems as well as in human societies and economies. Its commitment to IWRM replaces an earlier focus on localized sources of pollution and management of separate components of the ecosystem.

In 2003, the Water Convention was amended to allow accession by countries outside the UNECE region, thus inviting the rest of the world to use the Convention's legal framework and to benefit from its experience. The amendment entered into force on 6 February 2013, turning the Water Convention into a global legal framework for transboundary water cooperation. This is of particular importance for countries that border the UNECE region such as Afghanistan, China and the Islamic Republic of Iran, but also many countries in Africa, South and Latin America.

United Nations Watercourses Convention (UNWC) - United Nations Convention on the Law of the Non-Navigational Uses of International Water Courses
Conventions such as the UNECE facilitate additional joint transboundary efforts, for example the cooperation between Slovakia and Hungary. The UNWC entered into force on 17 August 2014. The UNWC is an overarching global legal framework that establishes basic standards and rules for cooperation between watercourse states on the use, management and protection of international watercourses. Irrespective of its unofficial status, the principles of the UNWC have been used to guide international and trans-boundary treaties and management.

While this convention is not explicitly water quality and ecosystem health focused, it does formulate the principles of shared water management such as equitable and reasonable utilization, participation, obligation not to cause significant harm, the principle of prior notification of planned measures and the protection, preservation and management of international watercourses (Part IV). It contains provisions on the protection and preservation of watercourse ecosystems; prevention, reduction and control of pollution; and consultations concerning the management of an international watercourse.

3 Overview of existing water quality guidelines and standards for freshwater ecosystems

3.2 Introduction

Background and purpose

This chapter provides a detailed review of existing water quality guidelines (WQGs) with a specific focus on ecosystem needs. In a large number of countries, the protection and rehabilitation of freshwater biota is a central part of the water policy. It is important to know which methods are used to assess the quality status of aquatic ecosystems and how quality objectives and standards are established. We also aim to determine which approaches are used to protect freshwater ecosystems and what role(s) WQGs play in improving the water quality. In addition, we sought to identify what enforcement mechanisms are needed and available for effective implementation of the WQGs. The answers to these questions are needed at various levels (national, international, transboundary, etc.).

The overall goal is to identify and review existing WQGs that protect the health of ecosystems and their respective mechanisms (institutional arrangements, processes, methodological approaches and reporting mechanisms).

The review will focus on the following subjects:

- Identification of existing WQGs or parts thereof which may be relevant for guidelines for freshwater ecosystems
- Analysis of WQGs for freshwater ecosystems which are most up- to- date, effective and innovative with special attention to the selected quality classes, indicators and water quality goals and criteria for ecological assessment
- Implementation phase and achievements of existing WQGs for freshwater ecosystems
- Experiences with platforms for implementation and enforcement, e.g. organizational and institutional structures and communication.

Recommendations will be made for establishing WQGs for ecosystems, for institutional arrangements and enforcement mechanisms based on the review and analysis of existing guidelines.

This review is based on publicly available documents and on publications available in English from 15 countries. It was attempted to obtain fairly even geographical distribution accounting for different climates and hydrological conditions.

Structure of the chapter

In »Section 3.2, water policies in a number of countries are analysed to determine whether protection of aquatic ecosystems is regulated and, if so, how. The analysis will focus on the subject of the WQG (human uses and/or ecosystems), selected indicators for assessment and classification of aquatic ecosystems, the legislative authorities (involved at national, federal, catchment levels), on the nature of the guideline (voluntary or mandatory) and public participation.

Sections »3.3 to »3.5 deal with the structural analysis of selected existing WQGs for ecosystems, which may provide frameworks and approaches for the development of international WQGs for ecosystems. Guidelines developed in Australia/New Zealand, the European Union (EU) and the United States (US) are reviewed. »Section 3.3 describes classes and categories used for the characterization of aquatic ecosystems. »Section 3.4 describes the main biological, physico-chemical and/or hydromorphological indicators employed. In »Section 3.5, numerical and narrative criteria for the ecological assessment of water bodies are reviewed together with integrated assessment methods. »Section 3.6 deals with the application of WQGs at basin level and general mechanisms for the implementation of WQGs. Summary comments based on these analyses are given in »Section 3.7.

3.3 Protection of ecosystem health in selected water quality guidelines

A comprehensive examination has been undertaken to compare existing WQGs. Water laws and WQG's of 15 countries or entities (such as the EU) were selected to be reviewed using the same guideline. Besides a short description of the main laws and guidelines concerning water quality policy for each country, the review focuses on the following items:

- Are the objectives human use and/or ecosystem oriented?
- What types of indicators are described to assess the water quality: physico-chemical, biological and/or hydromorphological indicators?
- Are water quality categories defined and how many categories are used?
- Which authority and/or management organization will implement the guidelines?
- Is following the WQG mandatory or voluntary?
- Is public participation an obligation by law?

Overview of existing water quality guidelines

Australia and New Zealand
The joint Australian and New Zealand Guidelines for Fresh and Marine Water Quality were established in 2000 (ANZECC/ARMCANZ, 2000a). The main objective was to provide an authoritative guide for setting water quality objectives required to sustain current or likely future environmental values (uses). In Volume 1, a framework for applying the guidelines is described. Further detailed guidelines are given for aquatic ecosystems, primary industries, recreational water quality and aesthetics, drinking water and their monitoring and assessment. Volume 2 (ANZECC/ARMCANZ, 2000b) provides the rationale and background information for the guidelines for aquatic ecosystems and Volume 3 deals with the rationale and background information concerning the primary industry (ANZECC/ARMCANZ, 2000c). The guidelines are not mandatory.

The National Water Quality Management Strategy (NWQMS) outlines a three-tiered approach to water quality management:

i. the national level for the establishment of a vision of achieving sustainable use,
ii. state or territory level implementation through state water planning and policy process and
iii. regional or catchment level for complementary planning, e.g. catchment strategies and implementation by relevant stakeholders.

It is stated that, ultimately, it is the responsibility of local stakeholders and the state or territory or regional government to agree on the level of protection to be applied to water bodies. The WQGs promote assessment that integrates biological and chemical monitoring of surface water and sediments. Procedures for deriving numerical trigger values for physical and chemical indicators are described and trigger values for those indicators are presented. Three ecosystem conditions are recognized: high conservation/ecological value systems, slightly to moderately disturbed systems and highly disturbed systems.

More recently, an aquatic ecosystem toolkit has been published for identifying High Ecological Value Aquatic Ecosystems (HEVAE) (Aquatic Ecosystems Task Group, 2012). The objectives are

- to provide a nationally coordinated approach to policy development for relevant cross-jurisdictional issues within the aquatic ecosystems context
- to develop a national framework for the identification and classification of HEVAE.

The Aquatic Ecosystems Toolkit consists of five modules.

1. National Guidelines for the Mapping of Wetlands (Aquatic Ecosystems) in Australia
2. The Interim Australian National Aquatic Ecosystems Classification Framework
3. Guidelines for Identifying HEVAE
4. Aquatic Ecosystem Delineation and Description Guidelines
5. The Integrated Ecological Condition Assessment (IECA) Framework (currently under development).

Whilst the Aquatic Ecosystems Toolkit is not designed to replace existing tools or systems for identifying and classifying potential aquatic ecological assets, it has been developed to complement and build on other systems.

Brazil

The National Water Resource Policy is established in Law no. 9433 (Brazil, 1997) and includes a National Water Resources Management System. The main objectives are to ensure the necessary access to water of an adequate quality, to ensure the rational and integrated use of water resources with a view to achieving sustainable development, and to prevent and protect against water crises due to either natural causes or the inappropriate use of natural resources.

The Law defines the river basin as the territorial unit for water resource planning. At institutional level, a new organizational framework is introduced to regulate the areas of competence at federal, state and river basin level. Water agencies may serve as the executive secretaries of the River Basin Committees. The National Water Agency (ANA) is legally liable for implementing the National Resources Management System and facilitating the creation of the river basin committees. In a detailed analysis of 15 years of Brazilian water resource management policy it is concluded that the new institutional framework, including among others river basin committees and water agencies, is in line with international trends and, despite the major progress that was made, the implementation process still faces many challenges, especially in the least developed regions of the country (Veiga and Magrini, 2013). Besides guidelines for drinking water quality, no other WQGs are available. An overview of freshwater quality in Brazil (ANA, 2012) shows that mainly physico-chemical indicators are used to describe the water quality. Also, indicators for the microbiological trophic state are used, e.g. the growth of algae. The diagnoses of the water quality are presented through indices: the Water Quality index (WQI), the Trophic State Index and the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI). The use of new indicators such as bioindicators and ecotoxicological tests is recommended and currently used in the states of São Paulo, Minas Gerais and Paraná (ANA, 2012).

ANA identified four classes of surface freshwater bodies according to their uses. The uses considered are visual amenity, navigation, livestock watering, irrigation, fishing, recreation, human consumption, aquaculture, and protecting and preservation of aquatic communities. A special class is defined for the preservation of aquatic communities and is mandatory for Conservation Units. A case study of the rehabilitation programme of the Upper Tietê River is presented in »Section 5.3.

Canada

The Canada Water Act (1985) contains provisions for formal consultation and agreements with the provinces. Within the federal government, over 20 departments and agencies have special responsibilities for freshwater. In 1985, a Federal Water Policy was released, which has given focus to the water-related activities of all federal departments and which will continue to provide a framework for action in the coming years as it evolves in the light of new issues and concerns. The overall objective of the Federal Water Policy is to encourage the use of freshwater in an efficient and equitable manner consistent with the social, economic and environmental needs of present and future generations. Part of the strategy is to encourage and support opportunities for public consultation and participation in integrated planning.

The Canadian WQGs and subsequent updates (CCME, 2014a) are national science-based goals for the quality of aquatic ecosystems. Guidelines are available for a number of uses and for the protection of aquatic life. Numerical guidelines for chemical substances are derived according to a general protocol (CCME, 2007). Factsheets concerning environmental toxicology and chemistry are presented for a large number of chemical pollutants (CCME, 2014b). Numerical guidelines are presented for the short term and for the long term, both for freshwater and marine aquatic ecosystems under the condition that sufficient data is available (CCME, 2014a,b). A WQI is presented as a tool for simplifying the reporting of water quality data. Once the CCME WQI value has been determined, water quality can be ranked by relating it to one of the following five categories: excellent, good, fair, marginal and poor (CCME, 2014c). Biological Indicators are not included in the index.

The numerical environmental quality guidelines are recommended values. Provincial and territorial jurisdictions may have or may develop their own science-based environmental assessment tools (e.g. criteria, guidelines, objectives and standards). The legislative authority for implementation of the national Water Quality Guidelines lies primarily with each provincial or territorial jurisdiction.

China

The Environmental Protection Law of the People’s Republic of China stipulates the objectives of water environmental preservation as “to ensure human health, maintain the effective use of water resources and the conservation of marine resources, maintain the ecological balance, and enhance the development of modern socialism”. The legislative framework for water quality includes a general Environmental Protection Law, put into force in 1989. Its 2014 revision became effective on 1 January 2015. The latest revision has made the law more stringent and included new measures for environmental protection and the Law on Prevention and Control of Water Pollution (1984) with the latest version in 2008. Surface and groundwater quality standards were established (China, 2002). The main water quality standards are Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD), nutrients and some heavy metals and organic contaminants. Water Quality Criteria (WQC) have been used in China for several years, mainly referring to the WQC system in the US (Zhen-guang et al., 2013). Some important WQC have been studied, including aquatic life, biological, sediment quality, lake nutrient and human health criteria. In the present phase the focus is on aquatic life criteria.

The state of the surface water quality is expressed in a range from Grade I to V. Physico-chemical WQC are established for each grade. The grades are described as follows:

> Grade I: Mainly for headstream and the national nature preserves
> Grade II: Mainly for drinking water resources in first-class protected areas, protected areas for precious fish, and spawning areas for fish and shrimp.
> Grade III: Mainly for drinking water resources in second-class protected areas, protected areas for fish and swimming areas.
> Grade IV: Mainly for industrial water resources and recreational use in which people do not contact water.
> Grade V: Mainly for agricultural water resources and water areas required for landscape.

The Ministry of Environmental Protection has a mission to prevent and control environmental pollution in the country through the overall supervision and coordination of environmental protection management. Provincial and municipal governments also play important roles in pollution control as they are responsible for environmental quality under their jurisdiction and they can apply more stringent local legislation and standards than the national government. China has already had a long history in river basin management. River basin authorities are under the jurisdiction of the Ministry of Water Resources. China has recently revised its law to include public participation.

China

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> Grade II: Mainly for drinking water resources in first-class protected areas, protected areas for precious fish, and spawning areas for fish and shrimp.

Grade III: Mainly for drinking water resources in second-class protected areas, protected areas for fish and swimming areas.
Grade IV: Mainly for industrial water resources and recreational use in which people do not contact water.
Grade V: Mainly for agricultural water resources and water areas required for landscape.

The Ministry of Environmental Protection has a mission to prevent and control environmental pollution in the country through the overall supervision and coordination of environmental protection management. Provincial and municipal governments also play important roles in pollution control as they are responsible for environmental quality under their jurisdiction and they can apply more stringent local legislation and standards than the national government. China has already had a long history in river basin management. River basin authorities are under the jurisdiction of the Ministry of Water Resources. China has recently revised its law to include public participation.

India
The Water Quality Assessment Authority (WQAA) was constituted under the Environment (Protection) Act in 1986 by the Central Government to standardize methods for water quality monitoring and to ensure the quality of data generated, including water quality management aspects.

In 2005, the WQAA decided that the data generated by different authorities on water quality should be used for the formulation of a water quality management plan to help restoration of water quality. The Central Pollution Control Board (CPCB, 2008a) published guidelines for water quality management plans. The guidelines present stepwise activities required for the formulation of an action plan to restore water quality including setting water quality objectives, source inventory, maintenance of sewage treatment plans, the options that may be considered for action and various water conservation measures and financing. The following beneficiaries were considered: local citizens, protection of environment, protection of public health, protection of water resources (water supply, irrigation and other uses), protection of industrial use, enhanced property values and enhanced tourism.

Earlier, five classes of surface waters in India were defined based on the use of the water:

A. Drinking water source
B. Outdoor bathing
C. Drinking water source after conventional treatment
D. Propagation of wildlife and fisheries
E. Irrigation, industrial cooling and controlled waste disposal

Quality criteria were set for the different classes. In a guideline for water quality monitoring (CPCB, 2008b), the indicators for the classification are mentioned (total coliform organisms, pH, Dissolved Oxygen (DO) and BOD) and criteria are set for these indicators.

Also, the 2008 guidelines for the national lake conservation plan (NRCD, 2008) were provided with the aim to prevent pollution from point sources by intercepting, diverting and treating the pollution loads entering the lakes. The interception and diversion works may include sewerage and sewage treatment for the entire lake catchment area. Public awareness, public participation and capacity building as well as training and research in the area of lake conservation are part of the activities mentioned in the guidelines.

Indonesia

A framework for water environment management in Indonesia has been stated under an amended law concerning environmental protection and management Law No. 32 of 2009. The aim of the Law is - among others – to assure human safety, health and life; assure the continuation of life of creatures and ecosystem conservation; preserve the conservation of environmental functions; and control the utilization of natural resources.

In the Law, 13 preventing instruments are established including environment quality standards. WQC are set as the benchmark for national water quality. The management of water quality and control of water pollution is regulated by the government order (decree) No. 82 of 2001 (MoE, 2001). The criteria are the minimum standards set by the national government. Decree No. 82 assigns standard values to 46 parameters in four classes, which are determined based on the type of water usage:

Class I: drinking water
Class II: raw water for recreation, fishery, animal husbandry, irrigation
Class III: fishery, animal husbandry, irrigation, industry
Class IV: irrigation, industry.

Indicators used to monitor the ecosystem listed in the water quality standards are mostly physical and chemical parameters. A different classification scheme has also been set based on the WQI calculation to classify the water bodies, especially rivers. Four classes have been set:

1. (Good);
2. (Slightly polluted);
3. (Polluted); and
4. (Heavily polluted) (MoE 2003).

For lake management, a draft guideline has been developed to evaluate the ecological status of national lake ecosystems (MoE 2011). Three classes have been set:

A. Good;
B. Disturbed; and
C. Damaged

Based on several of the following indicators: hydromorphology, trophic status, water quality, biodiversity, food web, eutrophication and carrying capacity (based on phosphor concentration). Management of water quality in Indonesia is divided within the three governmental structure levels:

a. Central (national/transboundary level);
b. Provincial level; and
c. District/city level.

This includes carrying out water quality monitoring. In 1986, the River Basin Development Authorities Act came into force.

The Indonesian WQG is currently clearly based on water use aspects. Indonesia does not have a WQG established for ecosystems yet, although there are attempts to develop such an ecosystem-focused guideline and a draft version already exists. As such, Indonesia is a suitable example of a country in transition from a rather utilitarian towards a more ecocentric approach in water management. In »Section 5.6 the development of WQGs in Indonesia and water quality management problem of a lake are presented in some detail.

Japan

As summarized by the Water Environment Partnership in Asia WEPA(MoEJ, 2012), the two main objectives of protecting the water environment in Japan are the protection of human health and the living environment. In order to achieve both objectives, environmental standards for ambient water quality have been established in the Basic Environment Law as the acceptable water quality levels that should be maintained in public waters and groundwater. There are two kinds of Environmental Quality Standards (EQS) for water: those for human health, which are uniform standards applicable to all public water bodies throughout the country; and those for conservation of the living environment (MoEJ, 1997). The EQS for human health include 27 toxic substances. The EQSs for conservation of the living environment include pH, BOD, suspended solids, DO and total coliform. For lakes, EQS are also established for total nitrogen and total phosphorus. The standard values are specified for different classes of water uses. Regulatory frameworks for ecological risk assessment and management of chemicals in Japan had been introduced since 2003 (Yamazaki, 2011). Yamazaki discusses the frameworks for different regulatory standards for the conservation of aquatic life and also discusses possible improvements to the protocol for deriving criteria for toxic pollutants for the conservation of aquatic life.

The Water Pollution Control Law, enacted in order to achieve the water quality targets, sets provisions for water quality conservation such as effluent regulations from factories and business establishments, ambient water quality monitoring, measurement standards for public water bodies and the total pollutant load control system, which are applied to all public water bodies (MoEJ, 2012).

In 1997, an amendment of the River Law inserted the “conservation and improvement of the river environment” as a principal goal. The River Law regards lakes as integral parts of the river system. The amendment also asked for strong public and stakeholder involvement. As described by Nakamura et al. (2006), river restoration was booming and, in the period from 1990 until 2005, 23,000 restoration projects were conducted. Following standardized protocols, nationwide baseline information on the ecosystem state of river corridors was gathered. This information includes data on fish, benthic invertebrates, plants, birds and other biota. River systems deemed important for the national economy and people's lives are designated as "Class A river systems" and administered by the Minister of Construction. The others are designated as "Class B river systems" and administered by the prefectural governors. A river basin approach is adopted. The River Bureau plans and implements a variety of projects to protect people from disasters caused by rivers, sediment, storm surge and other natural phenomena, and to ensure sufficient water resources to support affluent lifestyles and develop attractive waterside environments. The River Bureau also drafts laws, manages river administration, issues licenses for water use and maintains facilities for the proper management of rivers, sediment control, and coastal protection.

Kenya

The Water Act is the main piece of legislation that regulates the water sector in Kenya. The Water Act came into force in 2002. The Act has various objectives and the description of the roles of various actors and the definition of water rights. The Act introduced a number of new water management institutions such as the Water Resources Management Authority (WRMA) to manage and protect Kenya’s water resources and Catchment Area Advisory Committees to enable the public and communities to participate in managing water resources in each catchment and to support WRMA at regional level.

The Water Act of 2002 (Republic of Kenya, 2002) regulates that a national water resources management strategy should be developed that prescribes the principles, objectives, procedures and institutional arrangements for the management, protection, use, development, conservation, control of water resources and, in particular, for

- determining the requirements of the reserve for each water resource,
- classifying water resources and
- identifying areas which should be designated protected areas and ground water conservation areas.

In 2014, a new Water Bill was sent to the National Assembly. It seems that the basin approach will be strengthened by the establishment of Basin Water Resources Committees and the development of basin area water resources strategies. Water quality and ecosystem objectives and regulations are not formulated in the new Water Bill.

Environmental regulation in Kenya is carried out by the National Environment Management Authority (NEMA). NEMA was established under the Environmental Management and Coordination Act No. 8 of 1999 and became operational in July 2002. Its role is to promote the integration of environmental considerations into government policies, plans, programmes and projects. As regards the water sector in particular, NEMA is in charge of formulating water quality regulations. In the Environmental Management and Coordination, (Water Quality) Regulations of 2006, water quality standards are given for sources of domestic water, effluent discharges, water used for irrigation purposes and water used for recreational purposes. No criteria or standards are given for ecosystem quality.

Nigeria

The Federal Environmental Protection Agency (FEPA) was established in 1988 by the Federal Government of Nigeria (FGN, 1988). The FEPA has statutory responsibility for overall protection of the environment and its initial functions and priorities. The National Policy on the Environment was launched in 1989. The introduction of guidelines and standards was part of the implementation of the policy and the environmental pollution abatement strategy contained therein. The guidelines and standards relate to six areas of environmental pollution control:

- Effluent limitations
- Water quality for industrial water uses at point of intake
- Industrial emission limitations
- Noise exposure limitations
- Management of solid and hazardous wastes
- Pollution abatement in industries

In 1991, "Interim Guidelines and Standards for industrial effluent, gaseous emissions and noise limitations" were published (FGN, 1991). These guidelines provide a large number of effluent standards for industries, but no standards for the quality of surface waters. Classification of environmental pollutants to set effluent standards is based on several factors for example: toxicity, persistence, physico-chemical characteristics, etc. The environmental objectives and goals determined the mode of classification. However, in order to ensure that various categories of pollutants are considered, the 129 priority pollutants identified by the United States Environmental Protection Agency (US EPA) have been adopted by the Agency pending the availability of new scientific data locally.

The development of River Basin Organizations (RBO) in Nigeria was analysed by Adeoti (2010). His findings were that there is no water management structure at a lower (sub-basin) level and a management platform that incorporates non-governmental stakeholders is lacking.

The Russian Federation

In 2006, Russia re-wrote its water code (Russian Federation Water Code No. 174-03) to focus on integrated regional water management. The code's founding principles are that protection of water bodies (both surface water and groundwater) takes priority over use, that usage shall not harm the environment and that utilization be prioritized toward drinking and other domestic purposes (Simpson, 2007). Some of the code's innovations include its river basin approach, the introduction of integrated water basin management schemes and civil society involvement in decision-making.

In terms of water quality, the code sets Maximum Allowable Concentrations (MACs) of chemicals, nuclear substances, microorganisms and other water quality indices. These norms are developed by responsible federal executive authorities for each water basin. These standards are mandatory and their violation is penalized. The environmental water quality standards are ecosystem oriented. In monitoring water bodies, chemical, hydrological and biological indicators are elaborated. For many chemical indicators, threshold values are set (MACs). Values for key hydrologic indicators (e.g. water discharge) depend on the type of use of the water body and are discussed and determined at the special governmental commission where all stakeholders take part. For water bodies that are used for drinking water supply, special pollution prevention zones are established.

Five water quality classes are defined when assessing water quality in a particular water body or a part of it. A system of regulations and bans is established for sewage discharges, along with dumping and discharges of harmful substances. In addition, a monitoring system is established, organized at the water basin level, to provide for regular observations on water quality and quantity, regimes of water use, data processing and updating of a state water register. The state water register, to which free access is available, is a compilation of documentation on water bodies and water basins, water quality and quantity, water use, hydro-technical facilities, and water protection zones. It also assembles the agreements and decisions on water use.

Concerning public participation, in a review on the Volga Basin (CABRI-Volga, 2006), it concluded that public participation and initiative in environmental decision-making is on a lower level than in the EU and that insufficient coordination between stakeholders and their interests is a bottleneck in problem-solving. At the same time, although the Volga Revival Programme was closed (in 2004), it has been a unique experience in basin-wide coordination and some of its participatory approaches had been successfully tested in practice.

South Africa

The concepts of Resource Quality Objectives (RQOs) and Resource Quality were introduced by the National Water Act (NWA) of 1998 (DWAF, 1998). This Act regulates a large number of water quality issues, e.g. regarding national and catchment management strategies, classification systems, for water resource, pollution prevention and the use of water.

However, the South African WQGs were already published some years before, in 1996,. The guidelines consist of eight volumes:

a. Domestic Water,
b. Recreational Water,
c. Industrial Water,

d. Agricultural Water Use: Irrigation,
e. Agricultural Water Use: Livestock watering,
f. Agricultural Water Use: Aquaculture,
g. Aquatic Ecosystems and
h. Field Guide.

Target Water Quality Ranges (TWQRs) are derived for the different uses. An overview of TWQRs is presented in the Field Guide (DWAF, 1996b), but for a number of constituents no values could be derived for certain uses because of a lack of data. The guideline for aquatic ecosystems (DWAF, 1996a) provides TWQRs for four categories of physico-chemical constituents: toxic constituents (mainly inorganic and a few organic), system variables such as pH and DO, non-toxic inorganic such as Total Dissolved Solids (TDS) and Total Suspended Solids (TSS), and nutrients. The number of classes for ecosystem quality is limited to two: below or above the TWQR. The guideline does not include (narrative) biological quality objectives.

The NWA defines the catchment area as the basic geographic unit of water quality management. A Catchment Management Strategy must be established in accordance with the requirements of the National Water Resources Strategy for each of the 19 Water Management Areas (WMAs). The importance of public participation is emphasized by the National Environmental Management Act of 1998. In »Section 5.7 the development of the water law and subsequent WQGs in South Africa is presented in more detail.

United States of America

The objective of the Clean Water Act 1972 (US EPA, 2016a) (CWA, 1972[36]) is to restore and maintain the chemical, physical and biological integrity of the nation's waters. The Act establishes the basic structure for regulating discharges of pollutants into the waters of the United States and regulating quality standards for surface waters. The basis of the CWA was enacted in 1948 and was called the Federal Water Pollution Control Act, but the Act was significantly reorganized and expanded in 1972. The "CWA" became the Act's common name with the amendments in 1972. An overview of the law and the major amendments is published by Copeland (2010). Under the CWA, the US EPA has implemented pollution control programmes such as setting wastewater standards for industry. A set of nationally recommended WQC for the protection of aquatic life and human health in surface waters is summarized for approximately 150 pollutants. These values provide guidelines for states and tribes to adopt WQC (US EPA, 2016a). (Tribes are Native American Indian Tribes and Heritage Groups that are recognized by individual states for their various internal government purposes; furthermore, they are also referred to as states). States are required to compile lists of water bodies that do not fully support beneficial uses. They are then required to calculate a Total Maximum Daily Load (TMDL), which describes a value of the maximum amount of a pollutant that a body of water can receive while still meeting water quality standards (US EPA, 2015c).

The US EPA also provides technical support for states concerning the development of biological criteria and biological assessment programmes. In 1985, a document was published that provides guidelines for deriving numerical criteria for the protection of aquatic organisms (US EPA, 1985). In 1990 and 1992, documents were published that provide guidance for the development and implementation of narrative biological criteria as part of a new priority for the development of Biological WQC (US EPA, 1990 and 1992). More recent approaches for developing criteria/benchmarks are discussed in US EPA 2011b; Cormier and Suter, 2013; Cormier et al., 2013). In 2011 (US EPA, 2011b), a comprehensive Primer on Using Biological Assessments to Support Water Quality Management was published.

[36] http://www2.epa.gov/laws-regulations/summary-clean-water-act

Three tools are described therein:

i. Biological Assessment Program Review,
ii. The Biological Condition Gradient (BCG) and
iii. Stressor Identification (SI) and Causal Analysis/Diagnosis Decision System (CADDIS).

In 2013, a comprehensive biological assessment programme review was published (US EPA, 2013b).

It is the primary responsibility of states to prevent, reduce and eliminate pollution, and to plan the development and use (including restoration, preservation and enhancement) of land and water resources. Comprehensive pollution control plans have to be made for a basin or portions thereof.

The CWA requires public participation in the development, revision and enforcement of any regulation, standard, effluent limitation, plan or programme. It is also stated that, to the maximum extent possible, the procedures utilized for implementing the CWA shall encourage the drastic minimization of paperwork and interagency decision procedures. »Section 5.5 summarizes the experience with biological state assessment in the continental US.

Vietnam

The Law on Environmental Protection (1993, revised 2005) is the principal law on environmental protection including water and stipulates that the objective of environmental protection is to ensure social progress in order to achieve national sustainable development. Environmental standards are also stipulated under the Law. In addition to the environmental law, the Law on Water Resources (1998), the Land Law (2003) and the Biodiversity Law (2008) complete the national legislation related to water and environmental management (MoEJ, 2012).

Physico-chemical water quality standards are established. The main parameters are BOD, COD, DO, TSS, N, P and metals. Four classes of surface water quality standards are set:

Class A1: good for domestic water supply and other purposes in A2, B1 and B2

Class A2: good for domestic water supply, but suitable technology must be applied; conservation of aquatic life or other purposes in B1 and B2
Class B1: Good for irrigation or other purposes with demand for similar quality water or other purposes in B1 and B2
Class B2: Good for water transportation and other purposes with demand for low quality water.

The Ministry of Natural Resource and Environment (MoNRE) is responsible for the management of the quality and quantity of water resources. Under the MoNRE, the Vietnam Environment Administration was established in 2008 to strengthen institutional capacity to manage environmental issues. It is responsible for policy planning, monitoring of compliance and provision of guidance to local governments. In implementation, local governments play an important role in environmental management, but the MoNRE takes a leading role in the promotion of environmental conservation activities through the implementation of environmental regulations and the provision of guidance. »Table 3.1 provides an overview of water quality guidelines for each of the above-mentioned countries and jurisdictional regions.

Table 3.3.1: Overview of WQGs for each country or jurisdictional region

Countr(y/ies)	Main Law(s) or Water Quality Guideline(s)	Reference	Objectives		Main Indicators and Water Quality Criteria/ Standards			Number of water Quality Classes		Scale of Legislative Authority for Implementation				Mandatory	Voluntary	Public Participation
			Human uses	Ecosystem	Chemical	Biological	Hydromorphological	Uses classes	Ecosystem classes	National	State or Territory	Regional/ Municipality	Catchment			
Australia/ New Zealand	Australian and New Zealand Guidelines for Fresh and Marine Water Quality	ANZECC/ ARMCANZ (2000 a-c)	×	×	×	×			3		×		×		×	×
Brazil	Law no. 9433 of 1997	Law no. 9433 (1997)	×	×	×			4(5)	2	×	×		×	×		
Canada	Canadian Environmental Quality Guidelines	CCME (2014)c	×	×	×			5		×	×				×	×
China	Environmental Protection Law (1989); Law on Prevention and Control of Water Pollution (1984)	MoEJ (2012)	×	×	×			5		×		×	×	×		
Colombia	“Decreto 1594”	Colombia (1984)	×	×	×			×		×				×		
European Union	Water Framework Directive (2000/60/EC)	EC (2000 a)	×	×	×	×	×		3	×			×	×		×
India	Guidelines for Water Quality Management	CPCB (2008a)	×		×			5			×			×		
Indonesia	Decree No. 82 of 2001	MoE (2001)	×		×			4	3	×	×	×		×		
Japan	Basic Environment Law (1997) Water Pollution Law River Law (amendment 1997)	MoEJ (1997). MoEJ (2012)	×	×	×	×		4(5)	3(5)	×		×		×		×
Kenya	Water Act	Republic of Kenya (2002)	×		×					×			×			×
Nigeria	Federal environmental Protection Agency (FEPA)/ River Basins Development Authorities Act	FGN (1988)	×		×								×	×		
Russia	Russian Federation Water Code	Russian Federation Water Code No. 174-03 (2006)	×		×		×	5		×			×	×		×
South Africa	South African Water Quality Guidelines	DWAF (1996 a-b)	×	×	×											
US	Water Quality Standard Handbook	US EPA (1983)	×	×	×	×		3	6	×	×		×		×	×
Vietnam	Law on Environmental Protection (1993, revised 2005) Law on Water Ressources (1998)	MoEJ (2012)	×	×	×			4		×		×	×	×		

Summary comments on existing water quality guidelines

- In all countries reviewed, water laws and/or WQGs have been established to protect human uses and in most cases also to protect aquatic life. Most of the laws and guidelines date from the 1980s and 1990s. Some have been partially adapted in recent years.
- All governmental frameworks include guidelines for physical indicators and chemical substances. They also provide strategies for pollution preventions, and measures and/or regulations to prevent and reduce discharges of pollutants, although the number of pollutants considered varies widely.
- However, only a few laws or guidelines focus more explicitly on the protection of the freshwater ecosystems by developing specific guidelines, by using biological and hydromorphological indicators and by taking into account other stresses than chemical pollution.
- The EU WFD may be considered as a framework, which has the most detailed and itemized ecological objectives for different types of water bodies. The objectives are based on biological, hydromorphological and physico-chemical quality elements.
- In the last decade in the US, new methods have been introduced for biological assessment to support water quality management. Australia developed an aquatic ecosystems toolkit for identifying HEVAE. In some other countries, initiatives have been taken to develop guidelines for aquatic ecosystems, e.g. in Indonesia, or to add biological indicators to regular monitoring programmes, e.g. in Brazil.
- In the majority of the countries reviewed, the water basin approach has been incorporated in water laws. In those countries, RBOs' play a role in the implementation of Integrated Water Resources Management (IWRM), as also described by Priscolli (2006).
- The importance of public participation has generally been acknowledged and has been laid down in law in most of the countries reviewed.

3.4 Classes for the quality status of ecosystems

Among the guidelines reviewed the ones prepared by Australia/New Zealand, the EU and the US provide well documented and the most up-to-date and innovative approaches for integrated ecosystem assessment including biological, physico-chemical and hydromorphological indicators. This section analyses the water quality classes for ecosystems in these guidelines.

Quality classes for ecosystems in existing guidelines

Quality classes for ecosystems in Australia and New Zealand
In the joint Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ, 2000a), three levels of ecosystem conditions are recognized and defined:

High conservation/ecological value systems: effectively unmodified or other highly valued ecosystems, typically (but not always) occurring in national parks, conservation reserves or in remote and/or inaccessible locations.

Slightly to moderately disturbed systems: ecosystems in which aquatic biological diversity may have been adversely affected to a relatively small but measurable degree by human activity. The biological communities remain in a healthy condition and ecosystem integrity is largely retained. Typically, freshwater systems would have slightly to moderately cleared catchments and/or reasonably intact riparian vegetation.

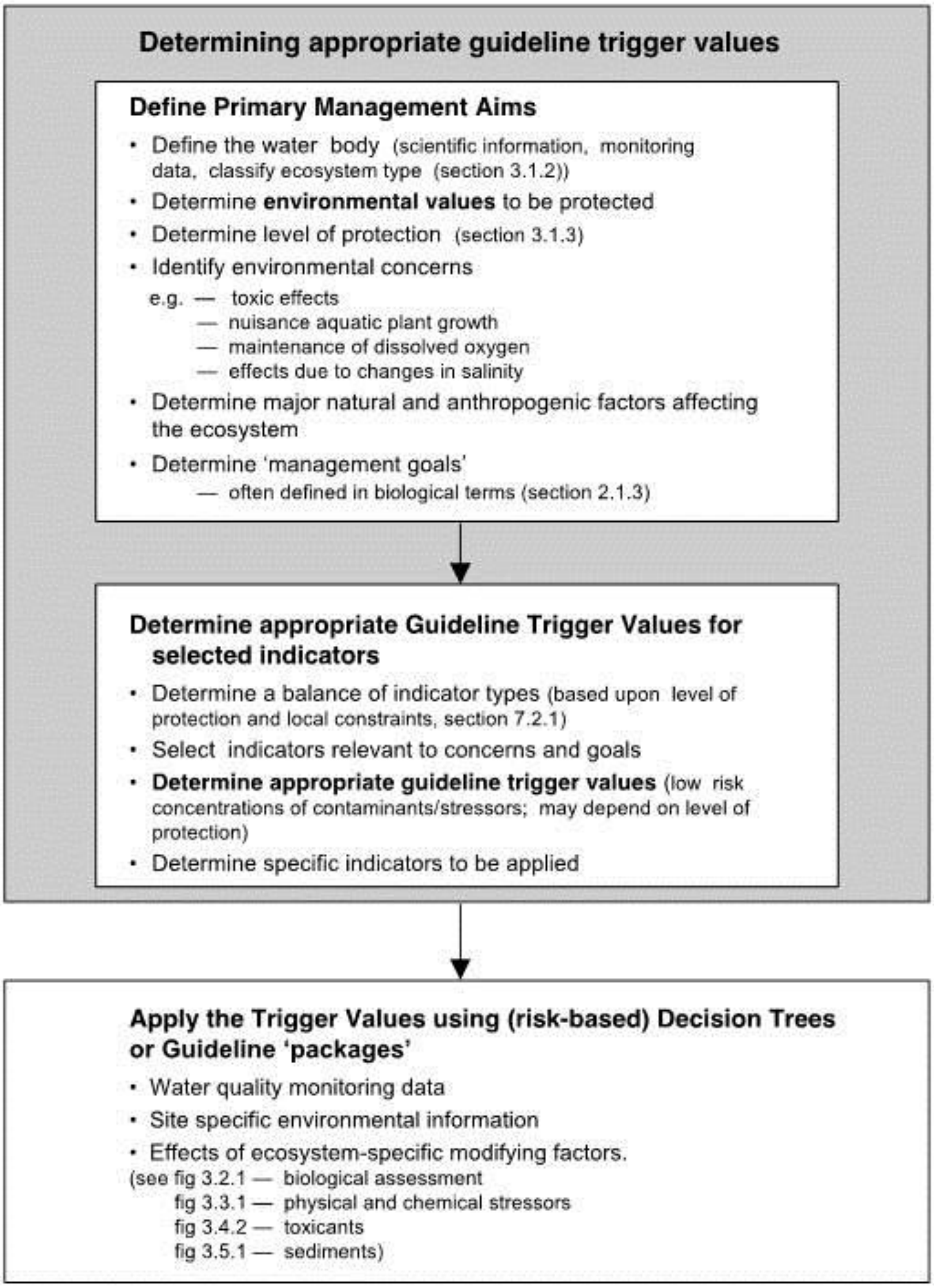
Highly disturbed systems: These are measurably degraded ecosystems of lower ecological value. Examples of highly disturbed systems would be some shipping ports and sections of harbours serving coastal cities, urban streams receiving road and storm water runoff or rural streams receiving runoff from intensive horticulture.

It is stated that the three levels should be considered as a practical but arbitrary approach to viewing the continuum of disturbance across ecosystems. The level of protection is the level of ecosystem quality desired by stakeholders and implied by the selected management goals and water quality objectives for the water resource. The valuation of a water body is part of the first step of the management framework of the guidelines as shown in »Figure 3.1. Stakeholders need to be actively involved in the steps. A number of examples of stakeholder involvement is presented in the guidelines.

The guidelines provide recommended levels of protection for each class using narrative descriptions of biological, physical and chemical stressors, toxicants and sediments. The guidelines recommend that the values of the indicators of biodiversity for the high conservation ecosystems should not change markedly. This means that any decision to relax the physical and chemical guidelines should only be made if it is known that such a degradation will not compromise the objective of maintaining biological diversity. For slightly and moderately disturbed systems, maintenance of their biological diversity condition to a suitable Reference Condition (RC) should be a key management goal. The third ecosystem class recognizes that degraded aquatic ecosystems still retain, or after rehabilitation may have, ecological or conservation values, but for practical reasons it may not be feasible to return them to a slightly – moderately disturbed condition.

For each level of protection, numerical trigger values for toxicants are presented in the guidelines. The highest protection level, 99% of the species expected to be protected, has been chosen as the default value for ecosystems with high conservation value. The 95% protection level should apply to slightly - moderately disturbed ecosystems. For biological indicators, and for physical and chemical stressors where no biological or ecological effects are available, the preferred approach to deriving guideline trigger values is from local reference data. For toxicants in water and sediments, the general trigger guideline values can be used, but data about reference sites may be a reason to change the trigger values.

Figure 3.1: Flow chart of the steps involved in applying the ANZEECC guidelines for protection of aquatic ecosystems. Figure and section reference numbers in the flowchart refer to the original document. Source: ANZECC/ARMCANZ (2000a).



Quality classes for ecosystems in the EU

As regarding ecological classification in the EU WFD (EC, 2000a), definitions are given for three classes of ecological status of water bodies. The general definitions are:
High status: There are no, or only very minor, anthropogenic alterations to the values of the physico-chemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions. The values of the Biological Quality Elements (BQEs) for the surface water body reflect those normally associated with that type under undisturbed conditions and show no, or only very minor, evidence of distortion.

Good status: The values of the BQEs for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions.
Moderate status: The values of the BQEs for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity and are significantly more disturbed than under conditions of good status.

Waters achieving a status below moderate shall be classified as poor or bad. No specific definitions are given for these classes. In the presentation of monitoring results, five classes are used for the classification of ecological status. Each class has a colour code ranging from blue, green/yellow, orange to red. The presentation of chemical monitoring results is limited to two classes: good (blue) or failing to achieve good (red). »Annex 4 of this document provides the detailed normative definitions of ecological status classification used in the EU WFD.

The general definitions for the high, good and moderate status are specified for the biological, hydromorphological and physico-chemical quality elements of rivers, lakes, transitional waters and coastal waters.
Member States shall protect, enhance and restore all bodies of surface waters with the aim of achieving good surface water status at the latest 15 years after the date of entry into force of the Directive in 2000. For heavily modified and artificial water bodies, their good ecological potential shall be achieved. The operational programmes of measures for achieving the aims shall be specified in river basin plans. The deadline may be extended for the purpose of the phased achievement of the objectives, provided that no further deterioration occurs.

Key elements of the monitoring such as the selection of monitoring sites and selection of substances, frequency of monitoring and standards for quality monitoring are described in the EU WFD. In order to ensure comparability of such monitoring systems, the results of the systems operated by each Member State shall be expressed as Ecological Quality Ratios (EQRs) for the purposes of the classification of ecological status. These ratios shall represent the relationship between the values of the biological parameters observed for a given body of surface water and the values for these parameters in the RCs applicable to that body. In »Section 3.5.3, more attention is paid to the quality ratios, intercalibration and the boundaries between classes.

Quality classes for ecosystems in the US

States of the US have used ecological classification systems for a long time. The procedure for initiating narrative biological criteria (US EPA, 1992) described how biological criteria can be used to help define the level of protection for “aquatic life use” within four hypothetical State-designated use categories ranging from class A (Highest quality or Special categories) to class D (Lowest quality water). The use of categories in policy-making and water management is the responsibility of states. All states use different methods and indices to determine biological condition, and therefore it is

difficult to determine if conditions vary across states and to develop national assessments, a descriptive model, the BCG has been developed (Davies and Jackson, 2006). The model shows an ecologically based relationship between stressors and the response of the aquatic community (»Figure 2.13 and »Figure A.6).
The US EPA adopted this method as one of the three tools for biological assessments (US EPA, 2011b). Six levels of biological conditions are described in the BCG (see »Figure 2.13). It provides a framework for understanding current conditions relative to natural, undisturbed conditions. The main purposes are to assess aquatic resources more uniformly and to communicate more clearly to the public. States are free in their use of the framework or in their adaption of the framework for their own water policy and water resource management. Nowadays, a number of states, among them Pennsylvania, use the BCG calibration.

National Recommended WQC are available for chemical pollutants (US EPA, 2016a). The Aquatic Life Criteria Table provides numerical values for more than 150 pollutants to protect aquatic life in fresh water and salt water. Two values are presented for both types of water. The Criteria Maximum Concentration (CMC) (acute) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect and is based on acute toxicity data. The Criteria Continuous Concentration (CCC) (chronic) is an estimate of the highest concentration of a material in surface water to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. Given these values, three “classes” of water quality may be distinguished: good quality (concentrations of pollutants in the water below CCC), moderate quality (concentrations between CCC and CMC) and quality at risk (concentrations above CMC). However, the US EPA has not published a federal system for such a classification. States use different methods to present the pollution status of their water bodies.

Summary comments on quality classes in existing guidelines

Quality classes for ecosystems are used for at least four reasons:

- to formulate present or future objectives concerning the status desired
- to present the ecosystem quality status in a transparent way ; awareness by uthorities and stakeholders
- to compare the quality status of different waters
- to report progress of the quality status.

- In the reviewed guidelines of Australia/ N¬¬ew Zealand, the EU and the US, three to six classes are used to describe the ecological condition of aquatic ecosystems. The highest class is always related to systems with unmodified and undisturbed status with a natural biological structure and functioning. The lowest class reflects highly disturbed systems with extreme changes in structure and function of the ecosystem.
- For each class narrative descriptions are given concerning the biological condition and level of disturbance. Methods and/or toolboxes are available to assess the ecological condition.
- Numerical values for concentration of toxicants are derived to guarantee a certain level of protection of the ecosystems. In Australia/New Zealand, these levels are directly related to the three classes of ecological condition. In the EU, two classes are distinguished concerning chemical quality: bad or good quality. In the US, two WQC are available: one based on chronic toxicity and the other on acute toxicity. These criteria are not applied to define water quality classes.
- The policy formulations concerning the distinct classes are rather different. In Australia/ New Zealand and the US, the classes form a framework for states and water management authorities to establish the ecosystem condition and to formulate aims and measures for preventing deterioration to improving the ecosystem condition. Member States of the EU shall protect, enhance and

restore all bodies of surface waters with the aim of achieving the “good surface water” class at the latest 15 years after the data of entry into force of the Directive in 2000. Under certain conditions, this deadline may be changed.

Indicators applied for ecological assessment

This Section deals with the biological, physical, chemical and hydromorphological indicators for ecological assessment and with the way these indicators are related to pressures and stressors. In this section, only the guidelines of Australia and New Zealand, the EU and the US are considered because they provide integrated approaches for ecosystem assessment including all the indicators mentioned.

Indicators applied for ecological assessment in Australia and New Zealand

The Australian and New Zealand Guidelines (ANZECC/ARMCANZ, 2000a) distinguish four types of guidelines and sets of indicators for ecosystem assessment:

- Biological assessment
- Physical and chemical stressors
- WQGs for toxicants
- Sediment quality guidelines

The guidelines for biological assessment are intended to detect important departures from a relatively natural, unpolluted or undisturbed state – RCs. The focus is on

- changes in species diversity, community composition and/or structure and
- changes in abundance and distribution of species of high conservation value or species important to the integrity of ecosystems.

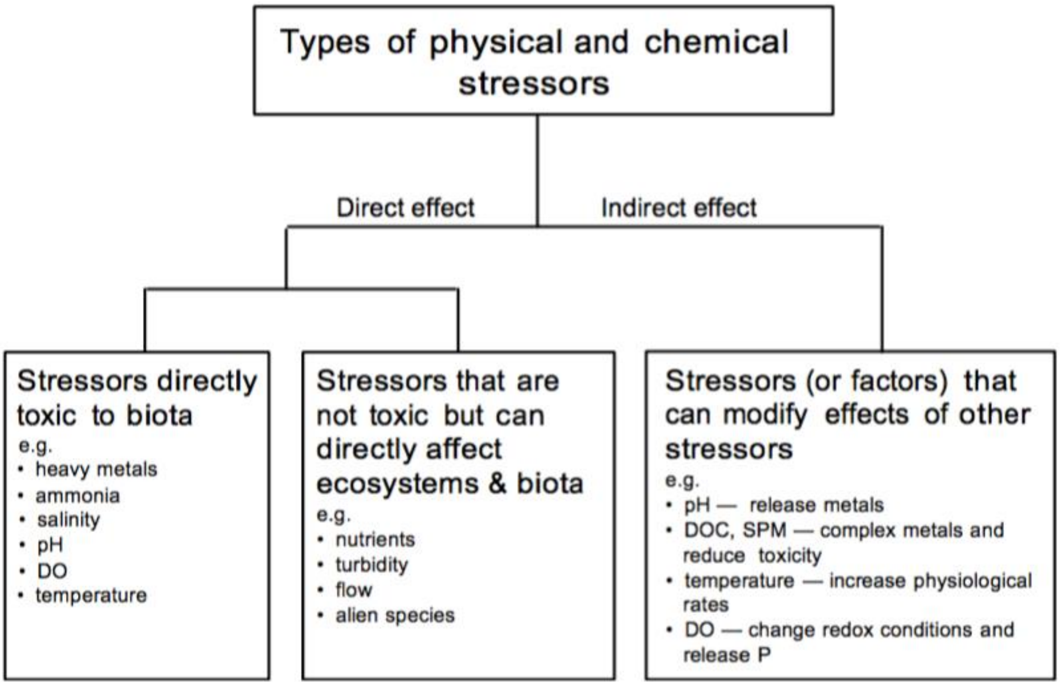
It is explained that bioassessment and biological indicators have come into use because the traditional physical and chemical guidelines are too simple to be meaningful for biological communities or processes.
To select the most appropriate biological indicators and protocols, three broad assessment objectives are described:

- broad-scale assessment (at catchment, regional or larger level),
- early detection of short- or longer-term changes and
- assessment of biodiversity.

For broad-scale assessment and early detection, Rapid Biological Assessment (RBA) methods are recommended because RBAs can be carried out at relatively low cost at a large number of sites or over a large geographical area. RBA based on stream macroinvertebrates is part of the Australian River Assessment System (AUSRIVAS).
A broad number of biological indicators may be used. The recommended biological indicators are related to the water quality issue. For example, if nutrient input might be the problem, the structure of phytoplankton or benthic algae communities and changes to vegetation structure are recommended as indicators for streams and wetlands. Other quality issues may require other indicators such as fish, macrophytes, zooplankton, frogs and aquatic and semi-aquatic reptiles and waterbirds (ANZECC/ARMCANZ, 2000b).

The physical and chemical stressors include a number of naturally occurring physical and chemical parameters, which can cause serious degradation of aquatic ecosystems when ambient values are too high and/or too low. The following indicators to assess the influence of stressors are considered: nutrients, biodegradable organic matter, DO, turbidity, Suspended Particulate Matter (SPM), temperature, salinity, pH and changes in flow regime. The effects of abnormal values may be direct or indirect, toxic or non-toxic. See »Figure 3.2.

Figure 3.2: Types of physical and chemical stressors and respective indicators. Source: ANZECC/ARMCANZ (2000a).



The WQG for toxicants provides trigger values for toxicants and explains how to use these. Trigger values are present for metals and metalloids, non-metallic inorganics and a large number of organic toxicants. See »Section 3.5 for how to derive them. The selection of indicators depends on the environmental concerns identified and the management aims formulated. See the flowchart of »Figure 3.1.

The establishment of guidelines for sediments will serve three principal purposes:

- to identify sediments where contaminant concentrations are likely to result in adverse effects on sediment ecological health;
- to facilitate decisions about the potential remobilization of contaminants into the water column and/or into aquatic food chains;
- to identify and enable protection of uncontaminated sediments.

Many urban and harbour sediments fall into the first category, usually being contaminated by heavy metals and hydrophobic organic compounds resulting from both diffuse and point-source inputs. Recommended guideline values for a range of metals, metalloids, organometallic and organic sediment contaminants are listed. The guideline numbers are trigger values that, if exceeded, prompt further action as defined by a decision tree (ANZECC/ARMCANZ, 2000a).

Besides the WQG, the Aquatic Ecosystems Toolkit provides indicators as already described in »Section 3.2. Whilst the toolkit has been developed for high ecological value ecosystems, it is not designed to replace existing tools or systems for identifying and classifying potential aquatic ecological assets; it has been developed to complement and build on other systems, and is flexible in its application. For the development of guidelines for ecosystems, it may be worth considering the criteria presented for the identification of HEVAE.

- Diversity: The aquatic ecosystem exhibits exceptional diversity of species (native/migratory), habitats and/or hydrogeomorphological features/processes.
- Distinctiveness: The aquatic ecosystem is rare/threatened or unusual; and/or the aquatic ecosystem supports rare/threatened/endemic species/communities/genetically unique populations; and/or the aquatic ecosystem exhibits rare or unusual geomorphological features/processes and/or environmental conditions.
- Vital habitat: An aquatic ecosystem provides vital habitat for flora and fauna species if it supports unusually large numbers of a particular native or migratory species and/or maintenance of populations of specific species at critical life cycle stages, and/or key/significant refugia for aquatic species that are dependent on the habitat, particularly at times of stress.
- Naturalness: The ecological character of the aquatic ecosystem is not adversely affected by modern human activity.
- Representativeness: The aquatic ecosystem is an outstanding example of an aquatic ecosystem class to which it has been assigned, within a drainage division.

Indicators applied for ecological assessment in the EU

The quality elements for the classification of ecological status as specified in the EU WFD (EC, 2000a) include three types of indicators:

- Biological elements
- Hydromorphological elements supporting the biological elements
- Chemical and physico-chemical elements supporting the biological elements

The biological elements include

- the composition and abundance of aquatic flora (phytoplankton, macrophytes and phytobenthos),
- the composition and abundance of benthic invertebrate fauna, and
- the composition, abundance and age structure of fish fauna.

Hydromorphological elements consist of indicators for the

- hydrological regime (quantity and dynamics of water flow, connection to groundwater bodies),
- river continuity (only for rivers) and
- morphological conditions (depth, structure and substrate of the bed, structure of the riparian zone).

The physico-chemical elements are divided into three groups:

- general elements (thermal conditions, oxygen conditions, salinity, acidification status, nutrient conditions),
- specific pollutants including all priority substances identified as being discharged into the body of water and
- pollution by other substances identified as being discharged in significant quantities into the body of water.

»Annex 4 of this document provides the full list of indicators used in the EU WFD. More details and implementation guides can be found in the following documents of the EC: EC, 2003; EC, 2005; EC, 2009 and EC, 2013.
In order to assess the magnitude of the stress to which bodies of surface water are subject, Member States shall monitor (surveillance monitoring) for those quality

elements which are indicative of the stressors to which the body or bodies are subject. In order to assess the impact of these stressors, Member States shall monitor as relevant:

- parameters indicative of the BQE, or elements, most sensitive to the stressors to which the water bodies are subject,
- all priority substances discharged and other pollutants discharged in significant quantities,
- parameters indicative of the hydromorphological quality element most sensitive to the stressor(s) /pressures identified.
-

In addition to surveillance monitoring, investigative monitoring shall be carried out: where the reason for any exceedances is unknown, where surveillance monitoring indicates that the objectives set out in Article 4 of the EU WFD for a water body are not likely to be achieved and operational monitoring has not already been established in order to ascertain the causes of a water body or water bodies failing to achieve the environmental objectives or to ascertain the magnitude and impacts of accidental pollution.

Concerning the identification of stressors, Member States shall collect and maintain information on the type and magnitude of the significant anthropogenic pressures to which the surface water bodies in each river basin district are liable to be subject, in particular the following:

- Estimation and identification of significant point-source pollution
- Estimation and identification of significant diffuse-source pollution
- Estimation and identification of significant water abstraction for urban, industrial, agricultural and other uses
- Estimation and identification of the impact of significant water flow regulation
- Identification of significant morphological alterations to water bodies
- Estimation and identification of other significant anthropogenic impacts on the status of surface waters
- Estimation of land use patterns, including identification of the main urban, industrial and agricultural areas, and, where relevant, fisheries and forests.

Member States shall carry out an assessment of the susceptibility of the surface water body status to the pressures and stressors identified above. Based on this analysis each Member State shall ensure the establishment for each river basin district, or for the part of an international river basin district within its territory, of a programme of measures in order to achieve the objectives established.

Indicators applied for ecological assessment in the US
Biological assessment is a principal monitoring tool and has been used to varying degrees and for various purposes by all 50 states over the past 20 years (US EPA, 2000c). The three major biological assemblages, or groups, monitored in comprehensive biological assessment programmes are fish, macroinvertebrates and algae. Monitoring of physical and chemical indicators has already been carried out for a much longer time according to the CWA of 1965, in which water quality standards became a feature of the law (Copeland, 2010). States were required to set standards and these would be used to determine actual pollution levels.
There are no federal lists for mandatory or recommended (assemblages of) indicators, but a lot of work has been done on tools for improving the use of biological assessment in the last few decades (US EPA, 2011b). Three tools are described:

- The Biological Assessment Program Review
- The BCG
- SI and CADDIS

A comprehensive report concerning the process of the Biological Assessment Program Review was published in 2013 (US EPA, 2013b). With the help of the programme review process described in the document, states can identify the technical capabilities and limitations of their biological assessment programmes and develop a plan to build on the programme's strengths and address its limitations. The document is intended to be used as a road map for the technical development of a biological assessment programme. It provides a step-by-step process for evaluating both the technical rigour of a water quality agency's biological assessment programme and the extent to which the water quality agency uses the information to support overall water quality management. The technical rigour of a biological assessment programme determines the degree of accuracy and precision in assessing biological condition and deriving stressor-response relationships. With increasing technical rigour, a water quality agency gains increased confidence in data analysis and interpretation as well as more comprehensive support for a variety of water quality management activities, including the following:

- More precisely defining goals for aquatic life and water use protection
- Deriving biological criteria
- Identifying high quality waters and establishing biological condition baselines
- Identifying waters that fail to support designated aquatic life uses
- Supporting development of WQC
- Conducting causal analysis
- Monitoring biological response to management actions

Four levels of technical programme rigour are distinguished (see Figure 3.3). A biological assessment programme's level of rigour is dependent on the quality and level of resolution of 13 technical elements and divided into three groups:

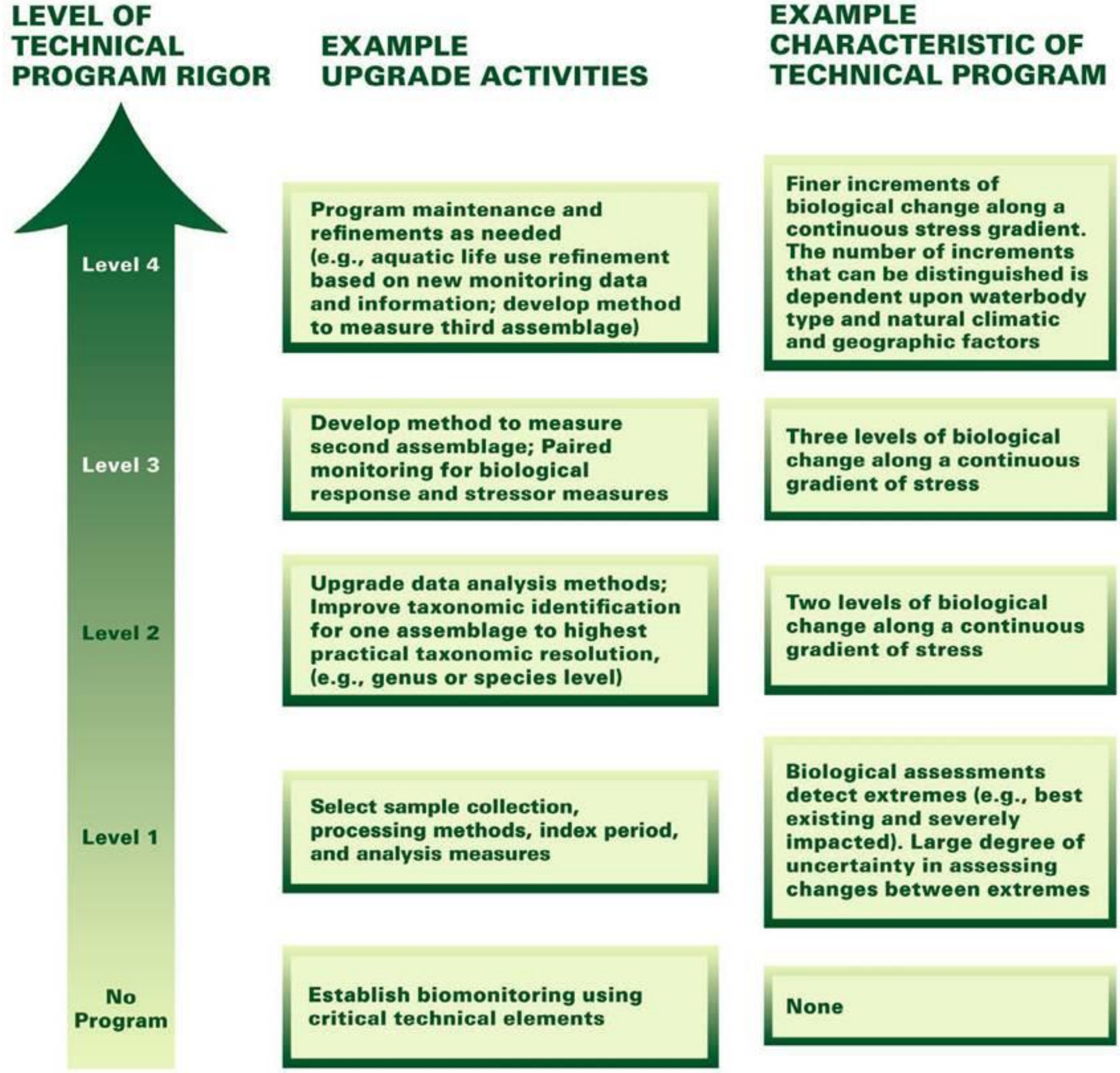
- Biological Assessment Design with the elements:
  - Index Period
  - Spatial Sampling Design
  - Natural Variability
  - Reference Site Selection
  - RCs
- Data Collection and Compilation with the elements:
  - Taxa and Taxonomic Resolution
  - Sample Collection
  - Sample Processing
  - Data Management
- Analysis and Interpretation with the elements:
  - Ecological Attributes
  - Discriminatory Capacity
  - Stressor Association
  - Professional Review.

The report describes the level of technical rigour for all of the 13 elements. The report also presents 10 biological and other ecological attributes to characterize the BCG. For example, highly sensitive taxa, intermediately sensitive and common taxa, and highly tolerant taxa are types of attributes, which are proposed.
The purpose of the SI and CADDIS is to identify the cause of biological impairment, which is established in the biological assessment. The core of this process consists of the following three main steps:

- Listing candidate causes of impairment
- Analysing new and previously existing data to generate evidence for each candidate cause
- Producing a causal characterization using the evidence generated to draw conclusions about the stressors that are most likely to have caused the impairment.

A comprehensive guidance document for SI is available (US EPA, 2000c). The Primer on Using Biological Assessments to Support Water Quality Management describes 17 case studies in different states and/or river basins (see »Section 5.5).

Figure 3.3: Example of typical upgrade activities which state or tribal water quality agencies have taken to incrementally strengthen their technical programmes. The example characteristics provided in column three are relevant to a biological assessment programme's technical capability to distinguish incremental biological change along a gradient of increasing stress. Improved ability to discriminate biological changes supports more detailed description of designated aquatic life uses and derivation of biological criteria. Source: US EPA (2013b).



Summary comments on indicators applied for ecological assessment

- Comprehensive guidelines and tools for ecological assessment are available and a lot of experience has already been gathered with their use.
- The most used biological indicators are composition and abundance of aquatic flora, macro invertebrates and fish. In some cases, species such as frogs and waterbirds are also used.
- The physico-chemical indicators can be divided into three groups:
  - general, naturally occurring elements such as nutrients, DO, pH and biodegradable organic matter
  - toxicants, and
  - other substances.
- Besides biological and physico-chemical indicators, hydromorphological indicators are relevant for assessing ecosystem quality.
- The selection of indicators is a process in which a number of elements is relevant. Most important are the type of water (lakes, rivers, wetlands, etc.), the management aims and the environmental concerns identified. In Europe, the indicators are to a large extent prescribed to establish whether the aquatic ecosystems have achieved a good quality status or not. Australia/ New Zealand and the US focus on guidelines for the states to select indicators as part of their ecological assessment programmes. For example, the US (US EPA, 2011b) published a comprehensive biological assessment programme including 13 elements and four levels of accuracy.
- Tools for analysing the results of the ecological assessment and the pressures and stressors, which may be the cause of impairment, are available. An example is the SI Guideline in the US.

Water quality criteria applied for ecological assessment

Definitions and terminology

Numerical values or narrative descriptions of indicators are needed to classify the ecological quality status of an aquatic ecosystem and can be used to set water quality objectives and standards.
Definitions: Several definitions and synonyms are used in WQGs. In “Water Pollution Control - A guide to the use of water quality principles“ (Helmer and Hespanol, 1997), published on behalf of the United Nations Environment Programme (UNEP), the Water Supply and Sanitation Collaborative Council (WSSCC) and the World Health Organisation (WHO), the following definitions are presented in »Chapter 3 of that publication by Enderlein et al. (1997):

- Water quality criterion (synonym: water quality guideline): numerical concentration or narrative statement recommended to support and maintain a designated water use;
- Water quality objective (synonyms: water quality goal or target): A numerical or narrative statement established to support and to protect the designated water uses of water at a specific site, river basin or parts(s) thereof; and
- Water quality standard: an objective that is recognized in enforceable environmental control at a government level.

New definitions and synonyms have subsequently been introduced in WQGs. In the Canadian WQGs (CCME, 1999), the terms criteria and guidelines are differentiated. Criteria are defined as scientific data evaluated to derive the recommended limits and guidelines as recommended numerical concentrations or narrative statements. In Australia and New Zealand (ANZECC/ARMCANZ, 2000a), the terms “WQG” and “Water quality objectives” are used according to the given definitions of Enderlein. The term

“Guideline trigger value” (and in the past “default value”) is used to indicate that, if exceeded, there is a potential to cause a problem and so it triggers a management response. In the US, the term “National recommended WQC” is used for numerical criteria for pollutants. Objectives and standards can be set by states and tribes. The EU WFD (EC, 2000a) defines EQSs: the concentration of a particular pollutant or group of pollutants in water, sediment or biota, which should not be exceeded in order to protect human health and the environment. These standards are also called chemical water quality standards. Member States shall take measures with the aim of achieving the EQSs of pollutants within 15 years of EU WFD´s entry into force.

At present, the terms “Threshold” or “Threshold value” are often used in ecological assessment, but different definitions are used. Firstly, a threshold value is the value of a key variable that will elicit a fundamental and irreversible change in the behaviour of the system. Groffman et al. (2006) define ecological thresholds as the points at which there is an abrupt change in an ecosystem quality, property or phenomenon, or where small changes in an environmental driver produce large responses in the ecosystem. On a general level, ecological thresholds are the breaking points of ecosystems at which pressures lead to abrupt changes in the ecosystem. Secondly, a threshold value is a value to delineate different classes or categories of water quality, e.g. the set of ecological standards along an ecological status scale (Irvine, 2012). The term threshold is frequently used in this sense in this volume.Another term regularly used for the classification of water quality is benchmark. A benchmark is defined as a chemical concentration or any similar attribute, specific to either water or sediment, above which there is the possibility of harm or risk to humans or animals in the environment. Benchmarks are meant to be used for screening purposes only; they are not regulatory standards. Benchmarks also serve as the identification of certain desirable levels, either to be achieved or to not fall below them (see »Section 2.7.1 and »Figure 2.12).

Criteria for biological indicators

As concluded in »Section 3.3, narrative descriptions are given for biological indicators to assess the biological status in a certain quality class in Australia and New Zealand, in the EU and in the US. This section will describe in more detail which (numerical) methods are applied to assess the status based on biological indicators and which approaches are applied to improve the comparability of the results of biological assessments of ecosystem categories.

Reference conditions

The value of biological indicators in aquatic ecosystems which are undisturbed (so-called RCs) is mostly both a starting point, and an important base for assessment of biological quality status. The Australian/New Zealand guidelines (ANZECC/ARMCANZ, 2000a) explicitly recommend that the preferred approach to derive guideline trigger values for biological indicators is from local reference data. Three sources of information are mentioned to define RCs:

I. historical data collected from the site being assessed;
II. spatial data collected from sites or areas nearby that are uninfluenced by disturbance; and
III. data from other sources if there is neither suitable historical data nor comparable reference sites nearby.

RCs in the EU WFD (EC, 2000a) are equated with the “high ecological status” of the classification system and are meant to represent the structure and functioning of biological communities under no or very minor anthropogenic disturbances. Member States should establish type-specific biological RCs representing the BQEs, which are prescribed in the guideline. For heavily modified and artificial water bodies, the relevant biological elements shall reflect, as far as possible, those associated with the closest comparable water body type. The European Commission (EC) published a guidance document concerning the typology, RCs and classification of rivers and lakes, which provides a common understanding of concepts and terms, and a stepwise approach for establishing RCs and ecological class boundaries (EC, 2003). »Annex 4 summarizes the classifications (typology) of water bodies adopted in the EU WFD.

The US EPA (US EPA, 2011b) recommends the use of information on the composition of a naturally occurring aquatic community to define goals for a water body. Many states have used such information to define more precisely their designated aquatic life uses, develop biological criteria, and measure the effectiveness of controls and management actions to achieve those uses. In the Biological Assessment Program review (US EPA, 2013b), knowledge about RCs is one of the key elements in a review. It is stated that the RCs serve as a benchmark for judging the conditions of a site and as a basis for derivation of biological criteria.

As the concept of “RCs” is increasingly being used to describe the standard or benchmark against which the current condition is compared, there is a need to bring some consistency to the use of the term (Stoddard et al., 2006). Stoddard et al. argued the need for a “RC” term that is reserved for the “naturalness” of the biota (structure and function) and that the naturalness implies the absence of significant human disturbance or alteration. They also propose terms for conditions which are different from the RC for biological integrity, e.g. Minimally Disturbed Condition (MDC), historical condition, Least Disturbed Condition (LDC) and Best Attainable Condition (BAC) and present a review of methods used for estimating RCs.

Deriving numerical criteria for biological indicators in Australia and New Zealand

Table 3.2: Division of AUSRIVAS O/E indices into bands or categories for reporting.
The names of the bands refer to the relationship of the index value to the reference condition (band A). For each index, the verbal interpretation of the band is stated first, followed by likely causes (bullet points). O/E represents the ratio of the number of families of invertebrates observed at a site to the number of families expected.
Source: ANZECC/ARMCANZ (2000a, Table 3.2.4 in referred document).

Band label	Band name	Comments	
		O/E Families	**O/E SIGNAL**
X	Richer than reference	More families found than expected Potential biodiversity 'hot-spot' Mild organic enrichment	Greater SIGNAL value than expected Potential biodiversity 'hot-spot' Differential loss of pollution tolerant taxa (potential disturbance related to water quality
A	Reference	Index value within range of central 80% of reference sites	Index value within range of central 80% of reference sites
B	Below reference	Fewer families than expected Potential disturbance either to water quality or habitat quality or both resulting in a loss of families	Lower SIGNAL value than expected Differential loss of pollution sensitive families Potential disturbance to water quality
C	Well below reference	Many fewer families than expected Loss of families due to substantial disturbance to water and/ or habitat quality	Much lower SIGNAL value than expected Most expected families that are sensitive to pollution have been lost Substantial disturbance to water quality
D	Impoverished	Few of the expected families remain severe disturbance	Very low SIGNAL value Only hard, pollution tolerant families remain

The Australian and New Zealand guidelines for Water Quality recommend the RBA for broad-scale assessment of biodiversity (ANZECC/ARMCANZ, 2000a). RBA procedures can be carried out at relatively low cost at a large number of sites or over large geographical areas. AUSRIVAS is an RBA method, which is often used in Australia (Linke et al., 2002). AUSRIVAS is based largely on River InVertebrate Prediction And Classification System (RIVPACS), which was developed in Britain and has been employed successfully using aquatic invertebrates. In AUSRIVAS, site data is compared with regionally relevant RCs. As a result of the more varied landscape, 48 models for individual states and distinct areas have been developed in order to achieve better resolution for assessing sites within a particular region (Simpson and Norris, 1997).

Two complementary indices summarize the outputs from the analysis of AUSRIVAS data:

i. the ratio of the number of families of invertebrates observed at a site to the number of families expected (O/E Family) and
ii. the ratio of the observed Stream Invertebrate Grade Number - Average Level (SIGNAL) and the expected SIGNAL value (ANZECC/ARMCANZ, 2000a).

The values of the indices are related to bands, which refer to the relationship of the index value and the RC (see »Table 3.2).

The more recently published guidelines for identifying HEVAE (Aquatic Ecosystems Task Group, 2012) do not provide numerical criteria. The narrative criteria are presented in »Section 3.4 of this chapter.

Deriving numerical criteria for biological indicators in the EU
The EU WFD recognizes the problem of comparability of biological monitoring results. To ensure comparability, the results of biological monitoring shall be expressed as EQR for the purposes of classification of ecological status. These ratios shall represent the relationship between the values of the biological parameters observed for a given y of surface water body and the values for these parameters in the RCs applicable to that body. The ratio shall be expressed as a numerical value between zero and one, with high ecological status represented by values close to one and bad ecological status by values close to zero. Each Member State shall divide the EQR for their monitoring system for each surface water category into five classes ranging from high to bad ecological status by assigning a numerical value to each of the boundaries between the classes. The value for the boundary between the classes of high and good status, and the value for the boundary between good and moderate status shall be established through an intercalibration exercise.

The following tentative scale of EQR values was established by a group of experts, based on their judgment of what would be appropriate intervals from high to bad in terms of species richness of benthic macroinvertebrates:

- High status: 1.00 – 0.80
- Good status: 0.80 – 0.60
- Moderate status: 0.60 – 0.40
- Poor status: 0.40 – 0.20
- Bad status: <0.20

The intercalibration process is aimed at ensuring comparability of the classification results of the EU WFD assessment methods developed by the Member States for the quality elements. The EC facilitates an exchange of information between Member States leading to the identification of a range of sites in each ecoregion in the Community. Geographical Intercalibration Groups (GIGs) have been established and a number of guidelines for the intercalibration has been published (EC, 2003; EC, 2011a). The process has been more time-consuming and methods are more complex than originally expected, but the development of assessment methods has been a transparent process and has resulted in improved and more standardized methods for assessing water bodies in Europe (Hering et al., 2010). For some BQEs or indicators, such as benthic invertebrates in coastal waters (Borja et al., 2007, 2009 cited by Hering et al., 2010) and phytoplankton biomass in lakes (as Chl-a), the intercalibration results were surprisingly clear: Most of the assessment systems give the same pattern. For other BQEs, such as phytoplankton composition in lakes, the first intercalibration results show large differences, e.g. in certain regions (Central - Baltic GIG). For some BQEs, such as fish, and one water category (transitional waters), the assessment systems had not been sufficiently developed to allow any intercalibration results in the first phase (2004 – 2008). Due to these shortcomings, the EC extended the intercalibration process with a second phase (2008 - 2011) to allow the completion of the intercalibration of all biological elements (EC, 2011a).

In an extensive overview of 297 biological assessment methods applied in Europe, Birk et al. (2012) found that the class boundary setting was mostly based on statistical principles (45%) and 37% of the assessment methods used ecological approaches alone or together with other approaches. In 18% of cases, class boundary setting was limited to expert judgments. They advocate better reflection of the necessary sampling effort and precision, full validation of stressors impacting relationships and an implementation of more ecological components into classification.

Deriving numerical criteria for biological indicators in the US

Numerical biological criteria have been developed by some states in the US. A few cases are described in the primer on using biological assessment (US EPA, 2011b). The Pennsylvania Department of Environmental Protection (PA DEP) has developed a new benthic macroinvertebrate Index of Biological Integrity (IBI) to assess the health of wadeable, freestone (high gradient, soft-water) streams. Additionally, PA DEP calibrated a benthic macroinvertebrate BCG and is exploring using the BCG to more precisely describe biological characteristics in Pennsylvania streams. Potentially, the BCG can be used in conjunction with the IBI to identify aquatic life impairments and to describe the biological characteristics of waters assigned special protection. The case description gives an example of the relation of the IBI score and the BCG level assignment (US EPA, 2011b).

Arizona has also developed numeric biological criteria to protect aquatic life and has established these values as water quality standards. On the basis of the statistical analysis of reference, stressed and test data sets, an attainment threshold of 25% of the reference site distribution was selected to be protective of aquatic life use. The non-attainment biological criteria threshold was set at the 10th percentile of reference, the level at which a majority of stressed samples occurs in the Arizona Department of Environmental Quality database. An inconclusive zone falls between the 10th and 25th percentile of reference. The zone of uncertainty encompasses variability in the Arizona IBI scores near 25%. To verify the biological integrity of the inconclusive samples, verification sampling is required before making an attainment decision.

In the US, the need for better comparability of the result of biological assessment is recognized as well. A national Wadeable Streams Assessment (WSA) has been carried out to evaluate the biological condition of streams in the US (US EPA, 2006b). Macroinvertebrate assemblages in each stream were analysed using a multimetric IBI and observed/expected indices were derived from the RIVPACS. Ultimately, 1,625 sites were selected and reference data was used to help to define nine large ecoregions. It is concluded that the WSA provided an unparalleled opportunity to push the limits of the conceptual and technical understanding of how best to apply the reference condition approach to the real world (Herlihy et al., 2008).

Criteria for general physical and chemical indicators

The common characteristics of general physical and chemical indicators are that these indicators also represent naturally occurring physical and chemical quality elements. The main indicators in this group concern oxygenation conditions, nutrient conditions, thermal conditions, transparency, acidification and salinity. When ambient values are too high or too low in comparison to RCs, serious degradation may be caused to the aquatic ecosystem. This section deals with the derivation of criteria in Australia/New Zealand, the EU and the US. See »Annex 2 for a comparison of the derived guideline values in a number of countries and entities.

Deriving numerical values for general physical and chemical indicators in Australia and New Zealand

For high conservation/ecological sites, the Australian and New Zealand guidelines (ANZECC/ ARMCANZ, 2000a) recommend that there should be no change from ambient conditions. For slightly or moderately disturbed systems, trigger values can be derived in terms of the 80th and/or 20th percentile values obtained from an appropriate reference system. For stressors that cause problems at high concentrations (e.g. nutrients, SPM, BOD, salinity), taking the 80th percentile of the reference distribution as the low-risk trigger value is recommended. For stressors that cause problems at low levels (e.g. low temperature water releases from reservoirs, low DO in water bodies), use the 20th percentile of the reference distribution as a low-risk trigger value. For stressors that cause problems at both high and low values (e.g. temperature, salinity, pH), the desired range for the median concentration is defined between the 20th percentile and 80th percentile of the reference distribution. Default trigger values have been derived for five geographical regions across Australia and New Zealand.

Deriving numerical values for general physical and chemical indicators in the EU

In the EU WFD, the ranges and levels established for the general physico-chemical quality elements must support the achievement of the values required for the BQEs at good status or good potential, as relevant. Since the values for the BQEs at good status will be type-specific, it is assumed that the ranges and levels established for the general physico-chemical quality elements should also be type-specific. All Member States shall derive general water quality standards and classify their water bodies in one of the five classes. For example, in the first phase, the United Kingdom (UK) published standards (related to certain biological element indicators) in rivers for BOD, DO (macroinvertebrates), ammonia (macroinvertebrates), pH (fish) and phosphorus (diatoms). For lakes, standards are given for DO (fish), conductivity (all species), acid neutralizing capacity (diatoms) and total phosphorus (phytoplankton biomass, macrophytes and phytobenthos). The derivation of a large number of additional standards is scheduled.

As regards eutrophication assessment, an extensive guideline document has been published (EC, 2009). Cardoso et al. (2009) published an overview of class boundaries based on average phytoplankton Chl-a concentrations for different types of lakes. Class boundaries for oligotrophic, mesotrophic, eutrophic, polytrophic and hypertrophic lakes are presented as well as for class boundaries of the EU WFD (reference, high/good and good/moderate boundaries).

In a review of the RBMPs of all Member States, it is concluded that standards have been set for some supporting physico-chemical and hydromorphological quality elements. However, most of the physico-chemical standards relate to nutrients and organic matter is, in most cases, not clearly linked to the good/moderate class boundaries for sensitive BQEs. If the programme of measures is based on nutrient standards that are too relaxed relative to the good/moderate boundaries for the BQEs, then good ecological status may not be achievable (EC, 2012).

Deriving numerical values for general physical and chemical indicators in the US
The nationally recommended Aquatic Life Criteria of the US EPA include the following general physical and chemical indicators: nutrients, oxygen, pH, suspended solids, and the transparency and temperature of water. Only narrative descriptions are available on national level for suspended solids and transparency; the criteria for the other indicators are described in documents taking ecoregional differences into account.

To address nitrogen/phosphorus pollution in rivers and streams, the US EPA recommends three types of scientifically defensible empirical approaches for setting numerical criteria (US EPA, 2000a), and in lakes and reservoirs (US EPA, 2000d): RC approaches, mechanistic modelling, and stressor-response analysis. More recently, a four-step process was developed for estimating and interpreting stressor-response relationships for deriving numerical criteria to address nitrogen/phosphorus pollution (US EPA, 2010b).

Criteria for toxic chemicals

The term toxicant is given to chemical contaminants such as metals, aromatic hydrocarbons, pesticides and herbicides. These toxicants are directly or indirectly discharged into aquatic ecosystems. The natural background concentrations are zero, except for metals. So, reference values cannot be used for the derivation of criteria. The common methods for deriving criteria are based on toxicity data from acute and chronic toxicity tests in laboratories and (semi-)field experiments and, to a lesser extent, specific field monitoring studies.

In Australia, the US and Europe over the last few decades, methods to derive criteria for toxic substances have been established and standards are set based on a growing amount of toxicity data. Most criteria relate to the concentration of a toxic substance in water, but for a number of toxic substances the concentration in sediment and biota may also be relevant to protect aquatic life. This is especially the case for those substances which, with very low water solubility, have a tendency to accumulate in the sediment and/or to bioaccumulate through the food web. If these substances pose a significant risk through indirect toxicity (i.e. secondary poisoning resulting from food-chain transfer) and their analysis is more feasible in other environmental matrices, such as biota and/or sediments, then a sediment or biota criterion may be required alongside, or instead of, the water column quality criterion.

A short overview of the methods applied in the Australia/New Zealand guidelines, the EU WFD and by the US EPA for deriving criteria for toxic substances is given in the subsequent sections. See »Annex 2 for a comparison of the derived guideline values in several countries and entities.

Numerical values for toxic substances in Australia and New Zealand
Most of the trigger values in the Australian/New Zealand guidelines (ANZECC/ARMCANZ, 2000a) have been derived using data from single-species toxicity tests on a range of test species because these formed the bulk of the concentration–response information. High reliability trigger values were calculated from chronic 'No Observable Effect Concentration' (NOEC) data. However, the majority of trigger values were moderate reliability trigger values, derived from short-term acute toxicity data (from tests ≤ 96 h duration) by applying acute-to-chronic conversion factors. As described by Warne (2001), two different methods were used to derive the guidelines: a modification of the Canadian assessment factor method and a new statistical distribution method called the Burr III method which was developed by Aldenberg and Slob (1993). The statistical distribution approach of Aldenberg and Slob has been adopted in the Netherlands and is recommended by the OECD (1992, 1995). The approach is based on calculations of a probability distribution of aquatic toxicity end-points. It attempts to

protect a pre-determined percentage of species, usually 95%, but enables quantitative alteration of protection levels. Volume 2 of the guidelines (ANZECC/ARMCANZ, 2000b) gives a very comprehensive description of the background to the approach. This includes, among others, the data used and incorporating bioaccumulation, bioconcentration, secondary poisoning, pH, hardness and other factors in the approach. The Australian/New Zealand guidelines (ANZECC/ARMCANZ, 2000a, 2000b) establish trigger values at four different protection levels: 99%, 95%, 90% and 80%. Here, the protection level signifies the percentage of species expected to be protected. The decision to apply a certain protection level to a specific ecosystem is the prerogative of each particular state jurisdiction or catchment manager, in consultation with the community and stakeholders. State jurisdictions or catchment managers can choose to apply different levels of protection to different ecosystem conditions if there is confidence that the disturbance is due to an overall physico-chemical disturbance and not just structural alteration.

Numerical values for toxic substances in the EU

The EU WFD (EC, 2000a) regulated that Member States shall derive all standards for toxic substances. However, the Directive was amended on this subject in 2008 (EC, 2008) and in 2013 (EC, 2013a). For a more effective regulation of surface water protection, it was decided that it is appropriate to set up EQSs for pollutants classified as priority substances at Community level and to leave it to Member States to lay down, where necessary, rules for the remaining pollutants at national level. The EU Directive of 2013 (EC, 2013a) provides EQSs for 48 pesticides, biocides (non-agricultural pesticides) and heavy metals as well as other groups of substances such as certain flame retardants. In Annex II of this EU Directive, EQSs are established as annual average concentrations and for some substances EQSs are also established for the maximum acceptable concentration. The EQSs are expressed as total concentrations in a whole water sample. For some very hydrophobic substances, e.g. mercury and hexachlorobenzene which accumulate in biota and which are hardly detectable in water, EQSs are set in biota.

Annex V, section 1.2.6. of the EU WFD (EC, 2000a) provides the procedure of setting chemical water quality standards by the Member States for the protection of aquatic biota. Standards may be set for water, sediment or biota. Where possible, both acute and chronic data shall be obtained for the taxa set out below which are relevant for the water body type concerned as well as any other aquatic taxa for which data is available. The base set of taxa are algae and/or macrophytes, daphnia or representative organisms for saline waters and fish.

To set a maximum annual average concentration, specific appropriate safety factors should be applied in each case consistent with the nature and quality of the available data should applied. Safety factors may vary:

- 1,000 if at least one acute L(E)C50 is present from each of three trophic levels of the base set;
- 100 if one chronic NOEC (either fish or daphnia or a representative organism for saline waters) is present;
- 50 if two chronic NOECs from species representing two trophic levels (fish and/or daphnia or a representative organism for saline waters and/or algae) are present; and
- 10 if chronic NOECs from at least three species (normally fish, daphnia or are representative organism for saline waters and algae) representing three trophic levels are present.

There are other cases, including field data or model ecosystems, which allow more precise safety factors to be calculated and applied on a case-by-case assessment basis. Where data on persistence and bioaccumulation is available, this shall be taken into account in deriving the final value of an EQS. The standards thus derived should be compared with any evidence from field studies. Where anomalies appear, the derivation

shall be reviewed to allow a more precise safety factor to be calculated. The standards shall be subject to peer review and public consultation allowing a more precise safety factor to be calculated.
Comprehensive technical guidance for deriving quality standards for toxic substances is available (EC, 2011). The guidance includes methods for deriving standards to protect water quality, biota standards (levels of toxicants in aquatic organisms) and standards to protect benthic (sediment dwelling) organisms. This guidance also states that all available data for any taxonomic group or species should be considered, provided the data meets quality requirements for relevance and reliability.

Numerical values for toxic substances in the US
The recommended WQC for aquatic life in the US include a list of approximately 60 substances; most of them are toxic pollutants. The criteria contain two expressions of allowable magnitude: a criterion maximum concentration (CMC) to protect against acute (short-term) effects; and a CCC. The criteria are derived for the total concentration of a toxicant in the water column. Only for heavy metals, the US EPA recommends the application of dissolved metal concentrations, which more closely approximate the bioavailable fraction.

The US guidelines for deriving numerical national WQC for the protection of aquatic organisms and their uses were already established in 1985 (US EPA, 1985). After a decision is made that a national criterion is needed for a particular material, all available information concerning toxicity to, and bioaccumulation by, aquatic organisms is collected and reviewed for acceptability. If enough acceptable data for 48- to 96-hour toxicity tests on aquatic plants and animals is available, this is used to derive the acute criterion. If sufficient data on the ratio of acute to chronic toxicity concentrations is available, this is used to derive the chronic or long-term exposure criteria. If justified, one or both of the criteria may be related to other water quality characteristics such as pH, temperature or hardness. The US EPA derives acute criteria from 48- to 96-hour tests of lethality. TheUS EPA derives chronic criteria from longer term (often greater than 28-day) tests that measure survival, growth or reproduction. Where appropriate, the calculated WQC may be lowered to be protective of commercially or recreationally important species.

The guideline also provides an approach for deriving sediment criteria. The equilibrium partitioning Sediment Quality Criteria (SQC) are the US EPA's best recommendation of the concentration of a substance in sediment that will not unacceptably affect benthic organisms or their uses.

The Water Quality Standards Regulation (US EPA, 2014a) allows states to develop numerical criteria or modify US EPA's recommended criteria to account for site-specific or other scientifically defensible factors. States may meet the requirements by choosing one of three scientifically and technically sound options (or some combination thereof):The recommended WQC for aquatic life in the US include a list of approximately 60 substances; most of them are toxic pollutants. The criteria contain two expressions of allowable magnitude: a criterion maximum concentration (CMC) to protect against acute (short-term) effects; and a CCC. The criteria are derived for the total concentration of a toxicant in the water column. Only for heavy metals, the US EPA recommends the application of dissolved metal concentrations, which more closely approximate the bioavailable fraction.

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1. adopt state-wide numeric criteria in state water quality standards for all toxic pollutants for which the US EPA has developed criteria guidance, regardless of whether the pollutants are known to be present;
2. adopt specific numeric criteria in state water quality standards for toxic pollutants as necessary to support designated uses where such pollutants are discharged or are present in the affected waters and could reasonably be expected to interfere with redesignated uses; and/or
3. adopt a "translator procedure" to be applied to a narrative water quality standard provision that prohibits toxicity in receiving waters. At a minimum, such criteria need to be developed for toxic pollutants, as necessary to support designated uses, where these pollutants are discharged or present in the affected waters and could reasonably be expected to interfere with designated uses.

The three options are discussed in more detail in the Water Quality Standards Handbook[37]. The state needs to demonstrate that its procedures for developing criteria, including translator methods, yield fully protective criteria for human health and for aquatic life. The US EPA's review process, which requires that criteria be based on sound scientific rationale and be protective of all designated uses, will proceed.

Criteria for hydromorphological indicators

The hydromorphological condition of an aquatic ecosystem can be characterized by its hydrologic regime (quantity and dynamics of water flow) and morphological conditions (depth, structure and substrate of the bed and riparian zones). The hydromorphological indicators are relevant to analyse the impact of hydromorphological changes on the functioning and structure of the biological community and to develop strategies for the recovery of a disturbed system.

Criteria for hydromorphological indicators in Australia and New Zealand
In the Australian/New Zealand guidelines (ANZECC/ARMCANZ, 2000a), the hydromorphological indicator “flow” is mentioned as one of the physical and chemical stressors. The factsheet “Environmental flows” for (ANZECC/ARMCANZ, 2000b) contains guidelines for the establishment of flow requirements needed to sustain the ecological values of rivers. As background, a brief summary is presented of the ecological effects that can be caused by changed flow regimes due to changes in the catchment, weirs and dams, and abstraction or diversion of water. A review is given of the methods that are currently in use for determining Environmental Flow Requirements.

[37] https://www.epa.gov/wqs-tech/water-quality-standards-handbook

As stated in the factsheet, a generic process for setting flow requirements is needed, since each river system will have different flow requirements and the publication of 'magic numbers' or 'rules of thumb' is not possible. There are still many unknowns associated with the setting of flow requirements, in particular the detailed relationships between flow and key ecological processes. Concerning future flow guidelines, Arthington et al. (2006) suggest that a region-by-region and country-by-country analysis using hydrological classification methods combined with ecological calibration could fairly rapidly provide global environmental flow guidelines within the coming decade. The development of scientifically credible flow management guidelines in distinctive physiographic and ecological regions of the world would make a major contribution to the resolution of conflicts over shared water resources and thereby help to ensure that societies continue to benefit from the biodiversity and essential ecological goods and services provided by river ecosystems.

The factsheet "Hydrodynamics" (ANZECC/ARMCANZ, 2000b) deals with the hydrodynamics in impounded waters (i.e. lakes, reservoirs, estuaries). Two indicators are mentioned:

i. Residence time of the water which may influence the growth of cyanobacteria. The recommended guideline is that residence times should be reduced to less than the average cell doubling time of the species of concern so that cells are flushed out of the system to prevent nuisance growths of cyanobacteria in standing water bodies.
ii. Thermal stratification which may occur in summer and may lead to dramatic physical, chemical and biological changes both in the upper layer as well as in the lower layer, i.e. anoxic conditions in the lower layer and releases of iron, manganese and nutrients to the upper layer.

Criteria for hydromorphological indicators in the EU

In the EU WFD, hydromorphological quality elements are required for the determination of high and good status. The values of the elements should reflect totally or nearly totally undisturbed conditions. For other status, classes the hydromorphological elements are required to have conditions consistent with the achievement of the biological elements. The guideline typology, RCs and classification systems (EC, 2003) list the hydromorphological elements supporting the biological elements (see »Table 3.3). For further reading see the implementation reports of the EU WFD[38]
Hydromorphological standards are less well developed than nutrient standards. Further developments are clearly needed, using available European Committee for Standardization standards for rivers and lakes habitat surveys as well as new research results and good examples from practice (EC, 2012). The REFORM project (Restoring Rivers for Catchment Management) is funded by the EC with the aim of providing a framework for improving the success of hydromorphological restoration measures to reach, in a cost-effective manner, the target ecological status or potential of rivers. A comprehensive review of ecological responses to hydromorphological degradation and restoration was published in 2013 (Wolter et al., 2013).

Table 3.3: Hydromorphological elements supporting the biological elements.
Left column refers to rivers, right column to lakes. Source: EC (2003, Table 2).

Rivers	Lakes
Quantity of dynamics of water flow	Quantity and dynamics of water flow
Connection to groundwater bodies	Residence time
River continuity	Connection to groundwater body
River depth and width variation	Lake depth variation

[38] http://ec.europa.eu/environment/water/water-framework/impl_reports.htm#third

Stucture and substrate of the river bed	Quantity, structure and substrate of the lake bed
Structure of the riparian zone	Structure of the lake shore

In relation to changes in hydromorphological conditions, it is important to note that Member States may designate a body of surface water as artificial or heavily modified when the changes to the hydromorphological characteristics of that body which would be necessary for achieving good ecological status would have significant adverse effects on, among others,

- the wider environment;
- navigation, including port facilities, or recreation;
- activities for the purposes of which water is stored, such as drinking-water supply, power generation or irrigation;
- water regulation, flood protection, land drainage; or
- other equally important sustainable human development.
- For these artificial and heavily modified waters, RCs are not applicable. These hydromorphological conditions are considered as a given condition on which the ecological potential should be established.

The moreblished Ecosystems Task Group, 2012) do not provide numerical criteria. The narrative criteria are presented in »Section 3.4 of this chapter.

Criteria for hydromorphological indicators in the US

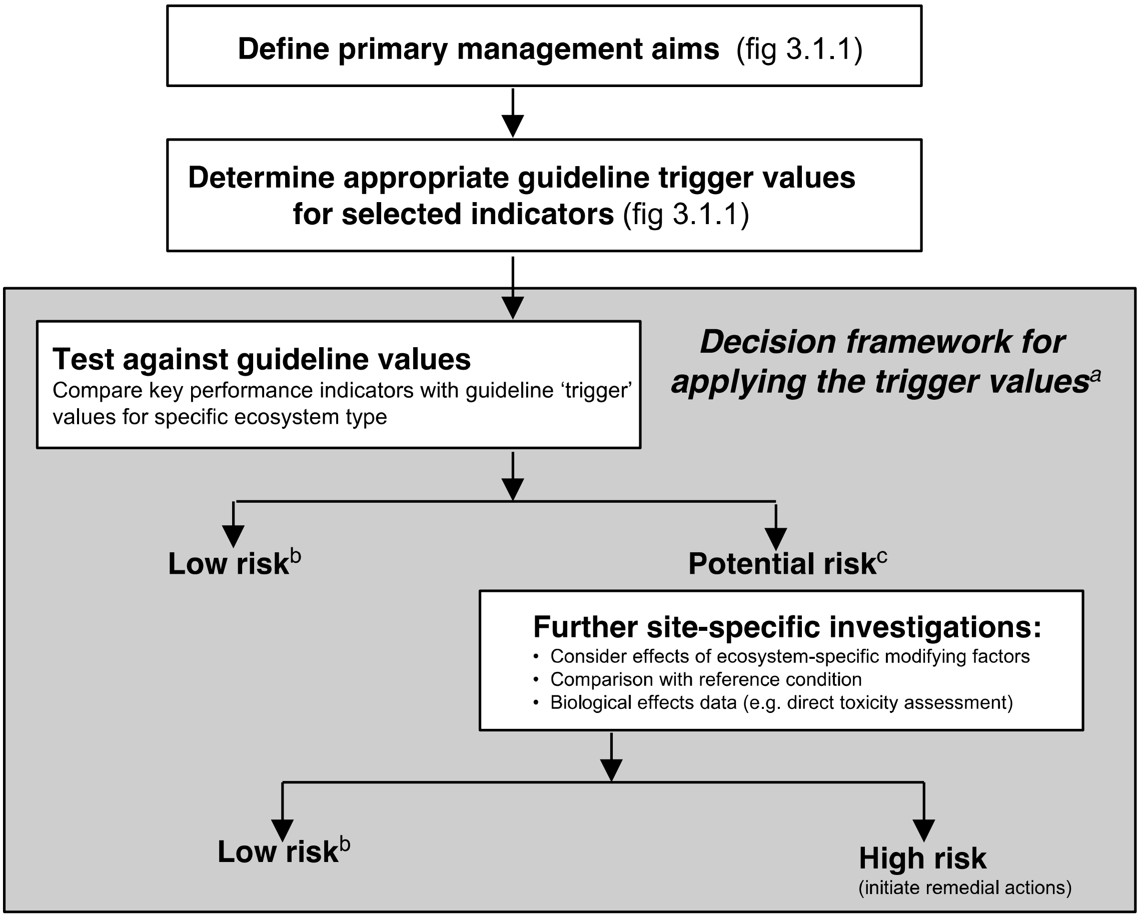
The changes of the physical habitat structure, such as sedimentation from stormwater runoff and physical habitat alterations from dredging, filling, and channelization, and changes in the flow regime are mentioned as stressors which may be analysed in biological assessments (US EPA, 2011b). At federal level, neither narrative nor numerical values are presented for these indicators, but a National Rivers and Streams Assessment 2008–2009 (US EPA, 2013c) was carried out in which – among others – four indicators of physical habitat condition were assessed: excess streambed sediments, riparian vegetative cover (vegetation in the land corridor surrounding the river or stream), riparian disturbance (human activities near the river or stream) and in-stream fish habitat. Stream conditions were also considered in this extensive study. Of these, poor riparian vegetative cover and high levels of riparian disturbance are the most widespread stressors, reported in 24% and 20% of the nation’s river and stream length respectively. However, excess levels of streambed sediments, reported in 15% of river and stream length, were found to have a somewhat greater impact on biological condition. Poor biological condition is 60% more likely in rivers and streams with excessive levels of streambed sediments (US EPA, 2013c).

Integrated ecological assessment

Integrated ecological assessment in Australia and New Zealand

The Australia/New Zealand guidelines provide decision trees for biological assessment, and assessing general water quality indicators and toxicants in ambient waters. If trigger values are exceeded, further site-specific investigations are recommended to examine whether the water quality is at low or high risk (»Figure 3.4). In the case of high risk, remedial actions should be initiated. The guidelines do not provide methods for integration of the results of biological, physical, chemical and hydromorphological indicator values. A comprehensive framework and guidance for the monitoring and reporting of fresh and marine waters and groundwater (ANZECC/ARMCANZ, 2000c) provide extensive methods for statistical data analysis and for reporting, but no methods are described on how to rank the information to make the results comparable with the results of assessments in other aquatic ecosystems.
The Aquatic Ecosystems Toolkit to guide the identification of HEVAE includes an IECA framework, but this module is currently under development.

Figure 3.4: Decision tree framework ('guideline packages') for assessing the physico-chemical stressors in ambient waters. The references in the flow chart refer to the original document. Source: ANZECC/ARMCANZ (2000a, Figure 3.3.1).



[a] *Local biological effects data and some types of reference data (section 3.1.5) generally not required in the decision trees*

[b] *Possible refinement of trigger value after regular monitoring (section 3.1.5)*

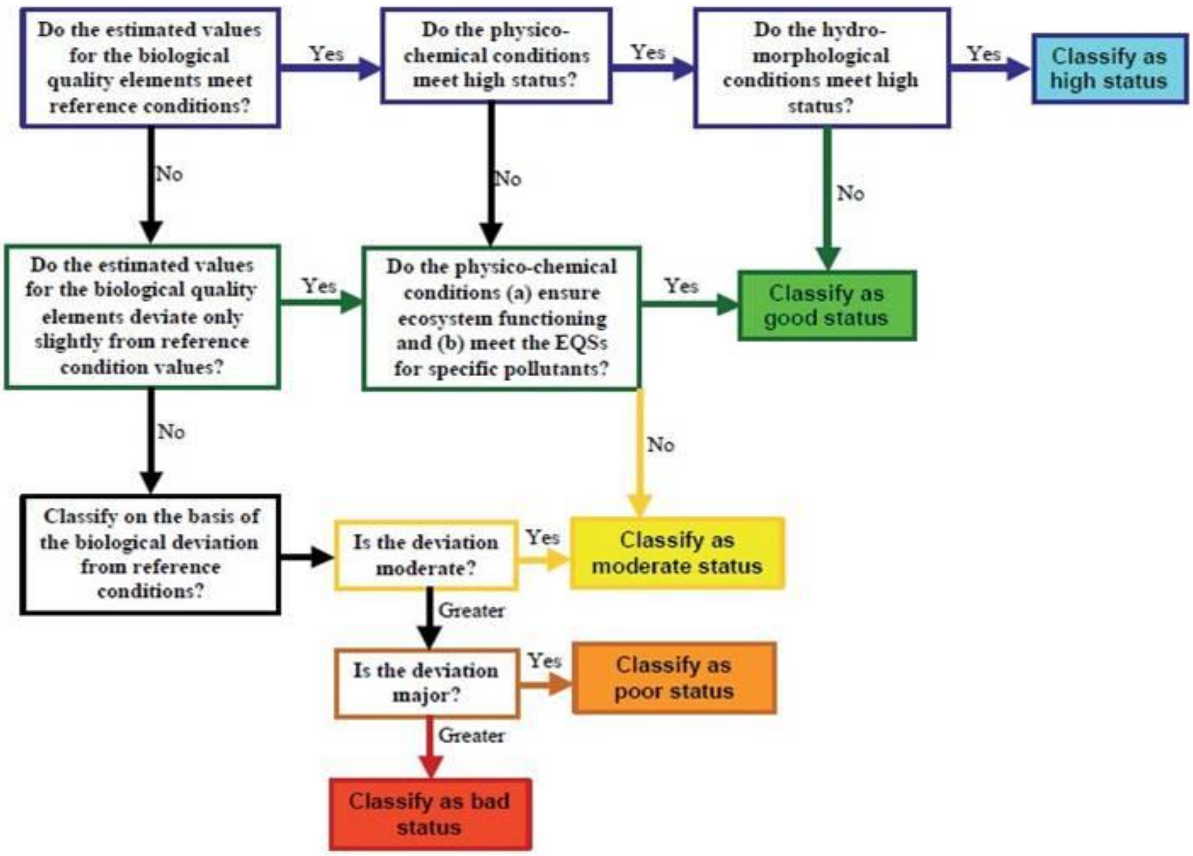
[c] *Further investigations are not mandatory; users may opt to proceed to management/remedial action*

Integrated ecological assessment in the EU

The EU WFD guidance on typology, RCs and classification systems (EC, 2003) presents a scheme concerning the relative roles of biological, hydromorphological and physico-chemical elements in status classification (see »Figure 3.5). The scheme deviates a little for highly modified water bodies and artificial water bodies because RCs are not available. The main reason for this guideline is to ensure comparability of the monitoring results of the systems operated by each Member State. In a recent overview of the status of the ecological status, based on the RBMPs, the results show that only around 44% of rivers and 33% of transitional waters are reported to be in high or good status. 56% of the lakes are reported to be in good or high status and 51% for coastal waters (EC, 2012).

A recent review of 252 EU WFD-compliant assessment systems published on http://www.wiser.eu/results/methods-db revealed that a large proportion (46%) of these systems target various forms of water pollution (acidification, eutrophication, heavy metals, pollution by organic compounds and pollution by organic matter). Other frequently addressed stress types are general degradation (19%), hydromorphological degradation (10%), habitat destruction (8%), riparian habitat alteration (5%), catchment land use (4%), flow modification (4%) and impact of alien species (4%), resulting in a higher diversity of stressors being assessed (Hering et al., 2010).

Figure 3.5: Indication of the relative roles of biological, hydromorphological and physico-chemical quality elements in ecological status classification according to the normative definitions in Annex V, 1.2 of the EU WFD. Source: EC (2003).

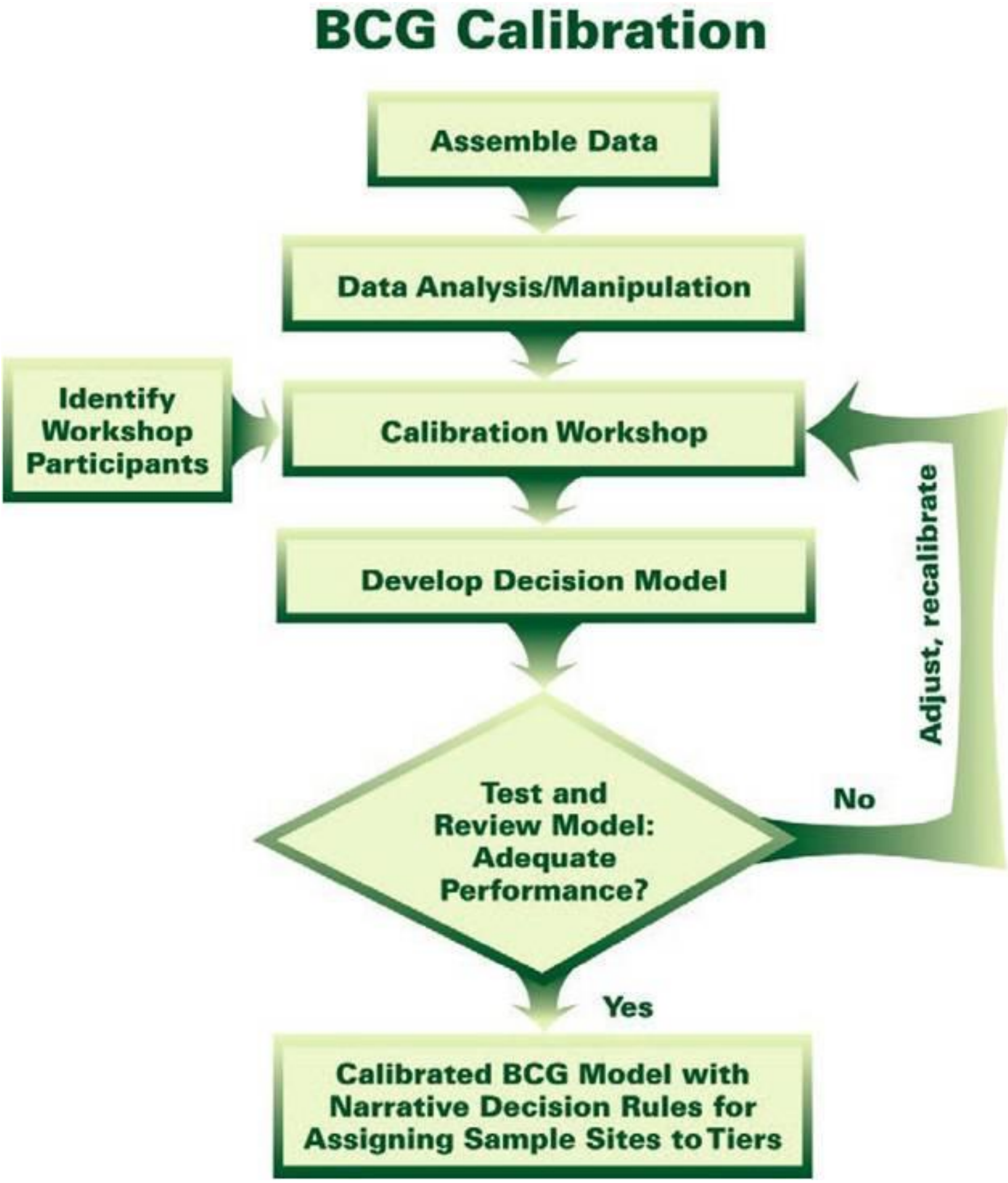


Integrated ecological assessment in the US

In the US, the BCG (see »Figure 2.13) was designed to provide a means to map different indicators on a common scale of biological condition to facilitate comparisons between programmes and across jurisdictional boundaries in the context of the CWA. The US EPA recommends this tool to describe how biological attributes of aquatic ecosystems change along a gradient of increasing anthropogenic stress (US EPA, 2011b). It provides a framework for understanding current conditions relative to natural, undisturbed conditions as described in »Section 4.5.5 (see Figure A 6). Some states, such as Maine and Ohio, have used a framework similar to the BCG to more precisely define their designated aquatic life uses.

It is a multistep process to calibrate a BCG to local conditions (»Figure 3.6). That process is followed to describe the native aquatic assemblages under natural conditions, identify the predominant regional stressors and describe the BCG, including the theoretical foundation and observed assemblage response to stressors.

Figure 3.6: Steps in a BCG calibration.Source: US EPA (2011c).



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Relations between biological indicators and stressors

In the three guidelines analysed in Sections 3.3, 3.4 and 3.5, a lot of effort is made to identify the cause of aquatic life impairments. In the following short review, some subjects will be highlighted.

Relations between biological indicators and stressors in Australia and New Zealand
In the Australian and New Zealand guidelines, two types of physical and chemical stressors that directly affect aquatic ecosystems are distinguished: those that are directly toxic to biota; and those that, while not directly toxic, can result in adverse changes to the ecosystem (e.g. to its biological diversity or its usefulness to humans). See also »Figure 3.1. Excessive amounts of direct-effect stressors cause problems, but some of the elements and compounds covered here are essential at low concentrations for the effective functioning of the biota — nutrients such as phosphorus and nitrogen, and heavy metals such as copper and zinc, for example. The guidelines provide a narrative description of the biological effects of unnatural changes in these general physical and chemical indicators, e.g. increasing levels of nutrients, lack of DO, excess SPM, unnatural change in salinity, in temperature or in pH. For toxicants, performing biological effects assessment is recommended if trigger values are exceeded and site-specific factors that may modify the guideline trigger values have been considered. The guidelines do not provide more detailed approaches for the identification of stressors. A site-specific problem analysis with the help of advanced assessment methods seems to be the course of action.

Relations between biological indicators and stressors in the EU
In the guideline concerning establishing RCs (EC, 2003), it is recommended that the use of both ecological and pressure criteria may be the most efficient way of screening potential reference sites or values or needs to aid in at least a preliminary assessment of the status of waters. Indeed, to establish RCs, it could be most cost-effective to start with stressor criteria because the reference community is defined as the biological community expected to occur where there is no or only very minor anthropogenic disturbance. In other words, to avoid circularity (i.e. the use of the same variable to delineate and validate the RC), stressor criteria may conveniently be used to screen for sites or values representing potential RCs. Once identified, biological elements should be used to corroborate this ecological high status. See »Figure 3.7. Uncertainty is a problematic issue in the first RBMPs in the assessment of ecological status (EC, 2012). There is no common understanding across Member States on how uncertainty should be assessed, and the information reported on uncertainty is often insufficient or missing in the RBMPs and associated documents. This lack of information especially concerns the uncertainty in the assessment methods themselves, e.g. uncertainty in relationships between the biological metrics used and the main pressures, as well as uncertainty in the boundary setting.

Relations between biological indicators and stressors in the US
Besides the Biological Assessment Program Review and the BCG, a third tool is recommended in the primer on using biological assessments (US EPA, 2011b): the SI and CADDIS. In 2000, the SI Guidance Document was published (US EPA, 2000c) with the intention of leading water resource managers through a formal and rigorous process that identifies stressors causing biological impairment in aquatic ecosystems and provides a structure for organizing the scientific evidence supporting the conclusions. The core of the SI process consists of the following three main steps:

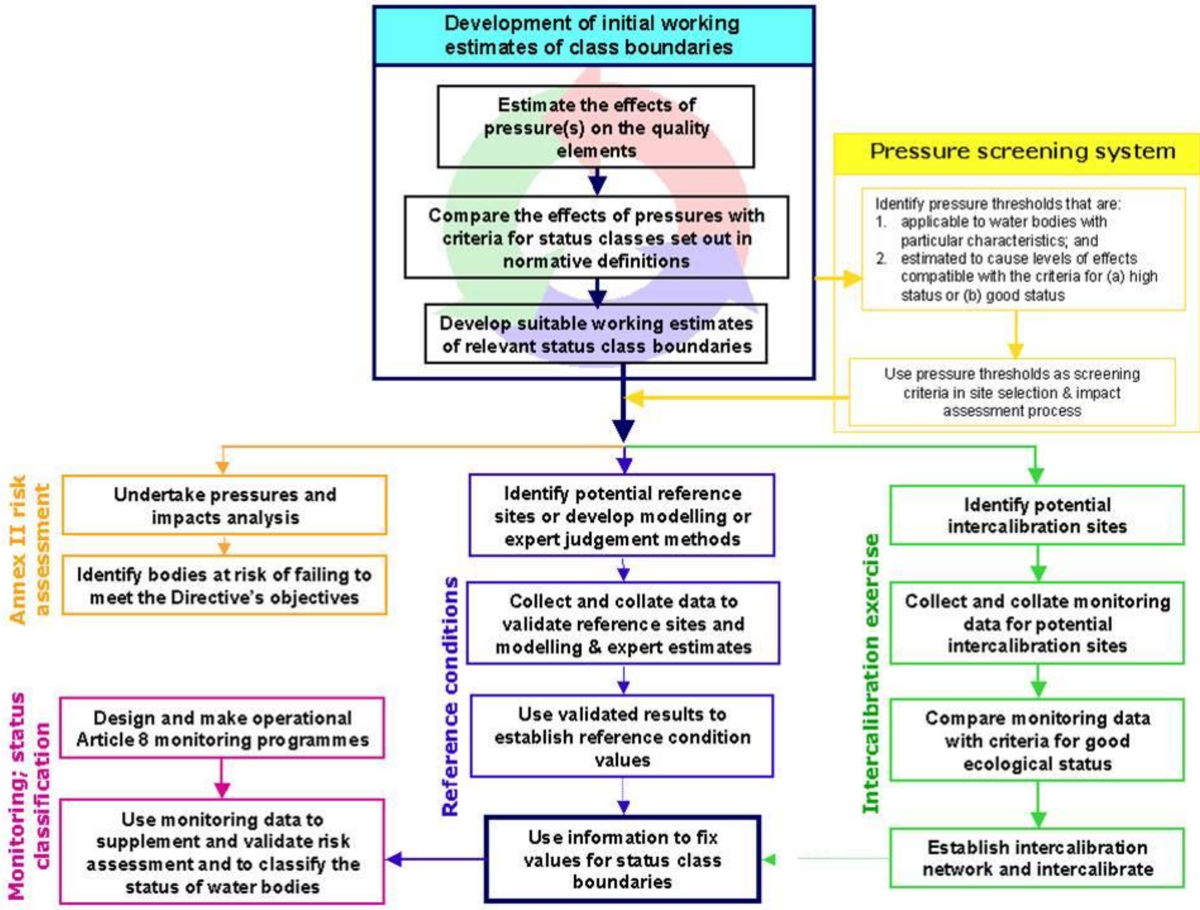
- Listing candidate causes of impairment.
- Analysing new and previously existing data to generate evidence for each candidate cause.
- Producing a causal characterization using the evidence generated to draw

conclusions about the stressors that are most likely to have caused the impairment.

The SI process is an iterative process as shown in »Figure 3.8. The kind of information needed includes information on the type of impairment, the extent of the impairment, any evidence of the usual causes of impairment (e.g. hydrological alteration, invasive species, habitat loss, toxicants, total nitrogen and phosphorus) and other information from the site. The evidence is considered first and then other, less direct kinds of evidence are gathered and evaluated, if needed. For example, one might consider other situations that are similar and can provide useful insights. CADDIS (Causal

Analysis/Diagnosis Decision Information System)[39] is an online application of the SI process that uses a step-by-step guide, worksheets, technical information and examples to help scientists and engineers find, access, organize, share and use environmental information to evaluate causes of biological effects observed in aquatic systems such as streams, lakes and estuaries. CADDIS also contains updates, clarifications and additional material developed since the SI guidance document was published (US EPA, 2000c).

Figure 3.7: The respective roles of pressure criteria and ecological criteria in identifying status classes.
Source: EC (2003).



Summary comments on water quality criteria applied for ecological assessment

- The definitions as given by Enderlein et al. (1997) still seem to be appropriate looking at the terms at present used in the guidelines considered. The definitions of Enderlein make a clear difference between recommended values (criteria or guidelines), established objectives and enforceable standards.

[39] http://www3.epa.gov/caddis/

- RCs play a major and increasing role in deriving biological criteria and in describing a standard or benchmark against which the current condition is compared. So, there is a need to bring some consistency to the use of the term. Stoddard et al. (2006) argued the need for a “RC” term that is reserved to the “naturalness” of the biota (structure and function) and that the naturalness implies the absence of significant human disturbance or alteration.
- Numerical criteria for biological indicators are nearly always related to RCs, e.g. the ratio of species observed for the site examined and known from reference sites.
- A large number of biological assessment methods is available and the number of comparisons of these methods is rapidly growing as well as the number of sites where these methods are applied.
- Guidelines for the quality assessment of naturally occurring physical and chemical quality elements such as oxygenation conditions, the nutrient condition, thermal conditions, transparency, acidification and salinity are available. Numerical criteria can be derived with the help of RCs. A huge amount of data are published especially on the impact of discharges of nutrients.
- Comprehensive guidelines are available for deriving criteria for toxic substances. It is clear that deriving WQC is a complex process of the integration of high-level scientific knowledge, taking into account a large number of uncertainties and policy definitions of protection levels. As criteria for toxic substances are in general not site-specific, except for heavy metals, it might be explored whether these criteria for toxic substances would be derived at international level as has been carried out, for example, for the WHO drinking water standards. So, the best knowledge in the world may be applied in deriving criteria and worldwide accepted criteria may play an important role in the protection of aquatic ecosystems.
- The hydromorphological condition of an aquatic ecosystem characterized by, for example, quantity and dynamics of water flow and morphological conditions such as depth, structure, substrate of the bed and riparian zones is relevant due to the impact of hydromorphological changes on the functioning and structure of the biological community and to develop strategies for the recovery of a disturbed system. Hydromorphological RCs are needed to develop general and site-specific criteria for hydromorphological assessment and restoration.
- Step-by step approaches and decision trees are developed to support ecological assessment based on biological, physico-chemical and hydromorphological quality indicators as well as methods for the classification of the status of an aquatic ecosystem.
- In the guidelines of Australia/New Zealand, the EU and the US, much effort has been made to develop methods for the identification of the cause of aquatic impairments and to clarify the relations between the biological structure and functioning, and the stressors, which may influence the biological structure and functioning. However, the presence of several stressors at the same time and the complexity of the ecosystem caused it to be difficult to prove the clear impact of certain stressors on aquatic life.

Application of water quality guidelines at basin level to protect freshwater ecosystems

WQGs need implementation mechanisms to achieve objectives established in the guidelines. The concept of the river basin as a unit of water management is widely accepted as an indispensable approach needed for IWRM. As the protection of fresh water ecosystems should be considered as part of IWRM, the basin concept is also the core for dealing with freshwater ecosystem health. Consequently, river (or lake) basin organizations can play an important role in supporting the implementation of WQGs for ecosystems. Moreover, a large number of aquatic ecosystems are transboundary systems and, consequently, protection of these ecosystems should include the whole catchment area. This also needs the cooperation of all countries sharing the basin.

Integrated water resource management and the role of (river) basin organizations

Recent developments have focused on an integrated river basin management, a subset of IWRM, and catchment management rather than single sector approaches. Key characteristics of sustainable river basin management are:

- Basin-wide planning to balance all user needs for water resources and to provide protection from water-related hazards;
- Wide public and stakeholder participation in decision-making, local empowerment;
- Effective demand management;
- Agreement on objectives within the basin and mechanisms for monitoring those agreements; and
- Adequate human and financial resources.

A team of international experts led by the World Wide Fund for Nature (WWF) and a Chinese team led by the General Institute of Water Resources and Hydropower Planning and Design (GIWP), Ministry of Water Resources, China drafted a book concerning river basin planning (Pegram et al., 2013). The book provides a comprehensive overview of strategic basin planning and techniques for basin planning. It mentions the following characteristics of the strategic approach to basin planning:

- trade-offs between economic, social and environmental objectives, and between existing and potential future demands;
- a sophisticated approach to recognizing environmental water needs and the importance of aquatic ecosystem functioning in providing goods and services;
- understanding basin interactions, including the range of hydrological, ecological, social and economic systems and activities at work at basin level; and
- robust scenario-based analysis to address uncertainty in future development and climate, by assessing alternative hydro-economic development, social justice and environmental protection.

It is stated that modern basin planning is increasingly developing ecological-based objectives, for example related to species and to ecosystems, rather than more traditional “environmental” objectives such as water quality objectives.

Varying opinions exist about the most effective scale of application: The success of a RBO may be dependent, for example, on the level of human and institutional capacity of the civil society, the degree to which water resources are developed and climatic variability. The policy and legislative framework will govern the purpose and effectiveness of the RBO. Generally, RBOs rarely have strong transnational law-making functions. A large number of publications concerning IWRM is available in the Toolbox of the Global Water Partnership[40]. Experience shows that all RBOs evolve with time and see their composition and duties adapted from time to time reflecting the real needs of the moment. GWP (2014) states that successful RBOs are supported by:

- An ability to establish trusted technical competencies;
- A focus on serious recurrent problems such as flooding or drought or supply shortages and the provision of solutions acceptable to all stakeholders;

[40] Global Water Partnership ToolBox (2013). http://www.gwp.org/en/ToolBox/

- A broad stakeholder involvement, catering for grassroots participation at a basin-wide level (e.g. through water forums);
- An ability to generate some form of sustaining revenue;
- The capacity to collect fees, and attract grants and/or loans; and
- Clear jurisdictional and appropriate powers.

An overview of RBOs, presented by Priscoli (2006), comprehensively describes the development of RBOs in the US, Canada, France, Germany, the Netherlands, Portugal, Great Britain, Spain, Russia, the Danube basin, Nigeria, Vietnam, China, Indonesia, Brazil, Mexico and Australia. Examples of RBOs, e.g. the Columbia River, Danube, Komadugu Yobe, Mekong and Yellow River, are also described. From the overview, it can be concluded that most of the RBOs are established to solve problems of (or similar to) flooding and droughts, to improve navigation and to manage hydropower stations. It became clear that these types of problems can only be solved on a basin scale, and cooperation between states and countries is needed to prevent or solve conflicts of interest. Water quality problems and ecosystem protection were, in most cases, not the main trigger for establishing RBOs. From the 1980s, the role of the river basin approach in tackling water quality problems increased.

Jaspers (2003) stated that water resources management on hydrological boundaries is not a new phenomenon, but the inability to manage water quality or to preserve environmental integrity and sustain environmental flows offered a new dimension.

Improving water quality and protecting ecosystems, and the role of basin organizations

Australia

The NWQMS (NWQMS, 1998) to improve water quality outlines a three-tiered approach to water quality management:

i.the national level for the establishment of a vision of achieving sustainable use;
ii.state or territory level implementation through state water planning and policy process; and
iii.regional or catchment level for complementary planning, e.g. catchment strategies and implementation by relevant stakeholders.

It is stated that, ultimately, it is the responsibility of local stakeholders and state or territory or regional government to agree on the level of protection to be applied to water bodies.

An independent evaluation of the national water strategy (KPMG, 2011) based on desktop analysis and stakeholder consultation found a number of shortcomings in the strategy, among others that the strategy does not have any specific vision, policy priorities or targets; that the updating of technical guidelines occurs on ad hoc basis; that the development time for technical guidelines is too long; that the technical documents are inconsistent in language and format; and that there are no performance metrics or reporting procedures in place to measure the ongoing effectiveness of the national water quality management. It is noted that implementation primarily occurs through various agencies, local councils, authorities and departments within each jurisdiction. The national guidelines are not mandatory and the policy framework and guideline application differ depending on the relevant structure and interlinked agencies and bodies in place in each state or territory. The evaluation does not provide any information about the role of basin organizations.

The best-known RBO in Australia is the Murray-Darling Basin Authority (MDBA). It is an independent expertise-based government agency responsible for the planning and management of both surface water and groundwater. In 2012, the Murray-Darling Basin Plan passed into law and has been a significant milestone in Australian water reform. The Basin Plan balances social, economic and environmental demands on the Basin's resources to ensure – among others – healthy and diverse ecosystems with rivers regularly connected to their creeks, billabongs and floodplains, and ultimately the ocean (MDBA, 2012). Furthermore, twelve case studies concerning the improvement of water quality are available with stories of progress and success from across Australia (Booth and Lamble, 2012). However, the study does not provide overall conclusions concerning results and success factors.

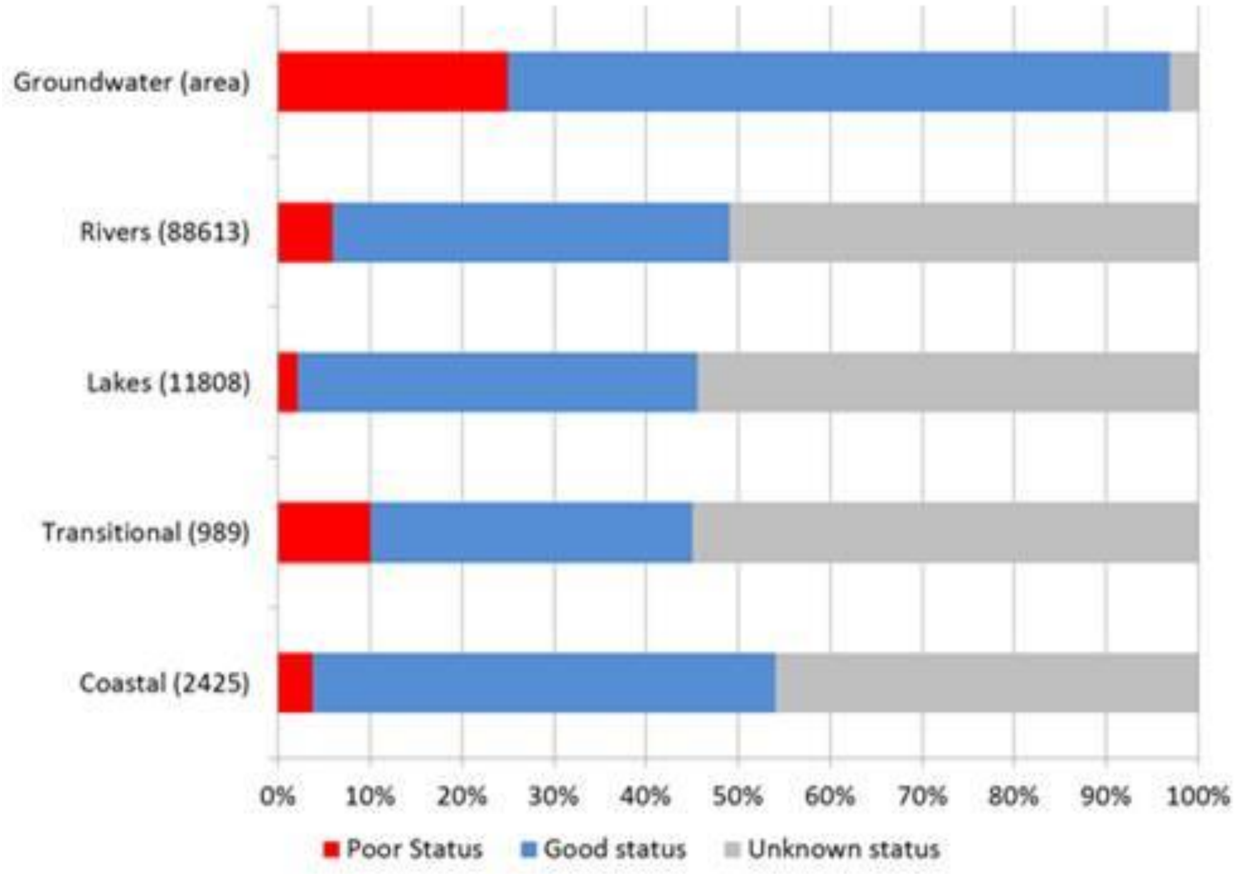
European Union

One of the most innovative aspects of the EU WFD is its river basin approach whereby water management is oriented based on hydrological, not political, boundaries (Moss, 2012). The River Basin District is the main unit for the management of river basins, which competent authorities need to identify to apply the rules of the Directive. There is a requirement to co-ordinate the actions (nationally and internationally) to achieve objectives established by the Directive. Member States shall ensure that a RBMP is produced for each river basin district. In the case of an international river basin district, Member States shall ensure coordination with the aim of producing a single international river basin plan.

All RBMPs are assessed in detail by the staff of the European Commission. The key aspects of the results of the assessment are reported in a so-called Commission's implementation report. The third implementation report was published in 2012 (EC, 2012). This comprehensive report provides, among others, the status and adoption of RBMPs, an overview of the status of EU waters and outlook, implementation of governance structures, classification of the ecological status and programmes of different kinds of measures. Some findings are:

- 121 RBMPs (out of a total of 170) have been reported;
- More than half (55%) of the total number of classified surface waters in Europe are reported to have less than good ecological status/potential;
- There is a high percentage of surface water bodies for which the reported chemical status is "unknown". See »Figure 3.9;
- There has been some progress in monitoring programmes since reporting to the Commission in 2007. For example, at EU level, there has been a 39% increase in monitoring sites in surface waters and 17% more for groundwater;
- In terms of transparency, it was found that the RBMPs from 11 countries (out of 25) were considered clear and well structured, whilst in some plans it was difficult to find the relevant information; and
- International cooperation has been significantly enhanced since the adoption of the EU WFD, in particular in some of the larger basins. International RBMPs have been adopted in catchments such as the larger Danube, Rhine, Elbe, Scheldt, Odra, Meuse and Ems.

Figure 3.4.1: Percentage of rivers, lakes, groundwater, transitional and coastal waters in good, poor and unknown chemical status in the EU. Source: EC (2012) and Water Information System for Europe (WISE).



United States of America
As described in several publications, e.g. Priscoli (2006) and Abdalla et al. (2010), the river basin approach has a long tradition in the US. To provide users with a comprehensive resource to develop more effective watershed plans as a means of improving and protecting the nation's water quality, US EPA published the Handbook for Developing Watershed Plans to Restore and Protect our Waters (US EPA, 2013d). The Handbook also provides guidance on how to incorporate the nine minimum elements from the CWA Section 319 Nonpoint Source Program's funding guidelines into the watershed plan development process. The nine elements from the CWA include the identification of causes and sources of pollution, estimation of pollutant loading, description of management measures, and identification and monitoring to measure progress. Since the Handbook was issued, the US EPA and other entities have stepped up watershed plan implementation, introduced new initiatives, developed new tools and provided additional funding sources.

The Handbook provides six steps in watershed planning and implementation processes:
1. Build partnerships
2. Characterize the watershed
3. Finalize goals and identify solutions
4. Design an implementation plan
5. Implement watershed plan
6. Measure progress and make adjustments

Restoration of aquatic ecosystems may be one of the goals of the watershed planning and implementation process. For example, a preliminary goal developed during the scoping phase, in step 1 of the watershed planning process, might have been to "restore aquatic habitat". Based on the information collected during data analysis, in step 2 of the watershed planning process, you might decide that the causes contributing to poor aquatic habitat including upland sediment erosion and delivery, streambank erosion, and near-stream land disturbance (e.g. livestock, construction) might be determined. Linking the preliminary goal to the source and impacts of pollution will be helpful to define the management objectives. In this case, appropriate management objectives could include (1) reducing sediment loads from upland sources and (2) improving riparian vegetation and limiting livestock access to stabilize streambanks (US EPA, 2013d).

As concluded in a collaborative survey during 2008-2009 (Draft, US EPA, 2013c), 21% of the nation's river and stream length is in good biological condition, 23% is in fair condition and 55% is in poor condition, based on a robust, commonly used index that combines different measures of the condition of aquatic benthic macroinvertebrates (aquatic insects and other creatures such as crayfish). Of the three major climatic regions (Eastern Highlands, Plains and Lowlands, and the West) discussed in this report, the West is in the best biological condition, with 42% of river and stream length in good condition. In the Eastern Highlands, 17% of river and stream length is in good condition; in the Plains and Lowlands, only 16% is rated in good condition.

Compliance and enforcement

For an effective process of the application of WQGs, compliance and legal mechanisms for enforcement are indispensable. Although basin organizations may play an important role in achieving compliance and in the enforcement of regulations, whether it will be effective strongly depends on the mandate, capacity and financing of the basin organization. General policies concerning compliance and enforcement will be presented in this section.

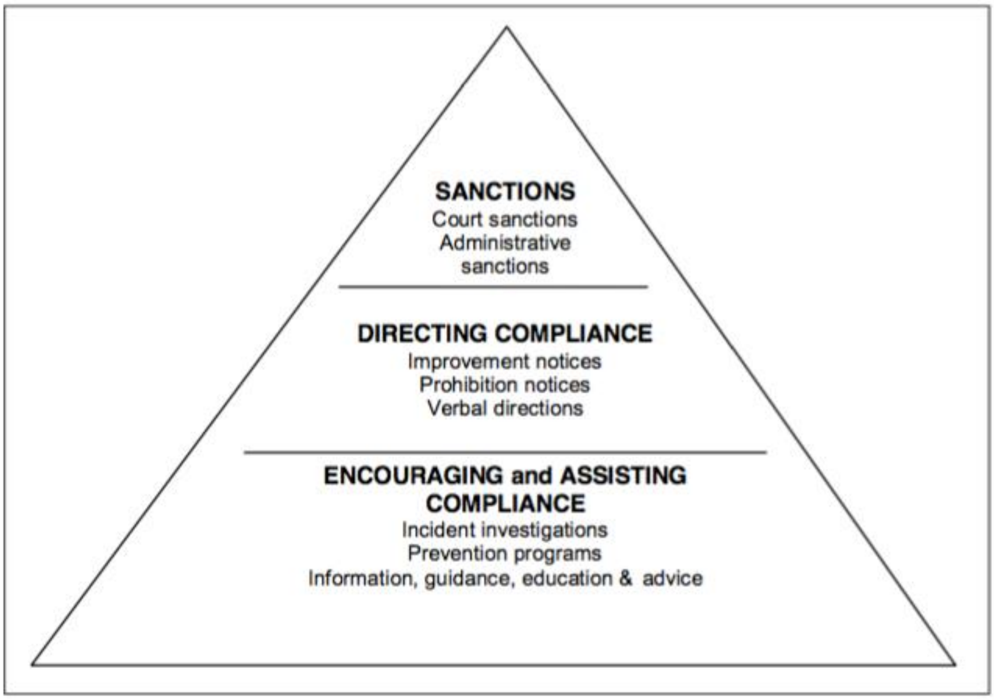
Compliance is defined as the full implementation of established requirements; it occurs when requirements are met and designed changes are achieved. Compliance is based on reliable monitoring and is conditional upon visible and effective surveillance, culminating in enforcement. Enforcement is the set of actions aimed at achieving compliance. This holds out the prospect of a society characterized by mutual respect and tolerance.

Promoting compliance is a matter that equally concerns those who make, implement and enforce policy and legislation. At national governmental level in most countries, policy directorates are responsible for developing and assuring the quality of a ministry's policy. Authorities or inspectorates are primarily responsible for enforcement and investigation. However, it is important, that policies are practicable and enforceable. Enforceability refers to the suitability of the legislation in terms of the ability of the competent authorities to use legal and administrative means at their disposal under domestic law to encourage or, in the event of wilful non-compliance, compel individuals to comply with their obligations under the legislation. Together, policy and enforcement must promote compliance by the public, companies and authorities themselves. Therefore, it is essential to agree on which rules must be assigned highest priority, how compliance can most effectively be achieved and where the compliance responsibilities lie. Such agreements require a shared view on the consequences of poor compliance and of the associated risks to society. The ultimate priority setting is obviously a political responsibility and not a civil service one.

Compliance and enforcement in Australia

The Australian National Framework for Compliance and Enforcement Systems for Water Resource Management (COAG, 2009) prescribes a risk-based approach to monitoring and enforcing compliance. The Framework defines a 'risk-based compliance strategy' as one that "identifies 'at risk' water resources and targets breaches of water resources legislation most likely to further stress the resource or which undermine the public's confidence in effective water resource management". The pyramid (see »Figure 3.10) is designed with most of the compliance actions at the base involving processes for encouraging and assisting compliance. Further up the pyramid, actions are more concerned with directing compliance through verbal directions, advisory notices and warning notices. The top, where generally there is the least activity, involves administrative remedies and criminal proceedings.

Figure 3.10: Enforcement Pyramid. Source: COAG (2012) and Ayres and Braithwaite (1992).



For the pyramid to work effectively, jurisdictions require each of the elements to be effective and operate efficiently to allow for the strategy's overall success. While these pyramids concentrate most resources at the bottom of the pyramid (for example, in educational programmes and technical assistance), the framework ensures that the tools and processes at all levels of the pyramid are equally robust. If any of the elements is weak, it opens a gap in the framework that can be exploited by those seeking to take advantage and can ultimately cause the failure of the whole approach.

The National Framework aims to provide a consistent approach by strengthening compliance and enforcement within each jurisdiction and addressing any gaps in their systems. This includes:

- robust compliance standards and enforcement strategies;
- rigorous and appropriate application of compliance standards and enforcement strategies;
- regular and consistent public reporting of monitoring and compliance action;
- raised public awareness; and
- an increase in resources to appropriate levels.

The State Offices in the States and Territories of Australia are responsible for managing the states' water resources. A key part of this responsibility is ensuring compliance with water management legislation to enable the secure and sustainable sharing of water between users. While most water users follow the rules and meet requirements, some people carry out illegal water activities. Water theft and harming a water source are serious crimes. These breaches can threaten water supplies for legitimate water users and harm the environment. The Compliance Policy (DSEWPC, 2009) explains how to prevent, detect and stop illegal water activities. This includes:

- assisting the community to understand their water rights and how to comply with the rules;
- monitoring water related activities to identify potential breaches; and investigating alleged breaches and taking appropriate action when a breach occurs.

The focus of the enforcement efforts is on the use of water, not on the ecological system.

Compliance and enforcement within the EU

Striking the right balance between flexibility in local implementation and robust and enforceable standards is essential for promoting adaptive capacity in water governance, yet achieving these goals simultaneously poses a unique difficulty. The Water Framework Directive is transposed into the national law of each Member State. National institutions therefore carry out enforcement.

The decentralized implementation of the EU WFD allows Member States flexibility in developing scale-specific water management policy and scale-specific solutions are crucial for adaptive governance (Green et al., 2013). The Directive provides flexibility for developing water policy at the appropriate level because geophysical circumstances differ per region (Keessen et al., 2010).

The structure of overlapping levels of control vary by Member State, as each state implements the EU WFD through different institutions, but all river basins plans are assessed, at the highest level, by the EU. See, for example, the third implementation report (EC, 2012). Under the provisions of the EU WFD, a Member State may create a new state-wide water management agency, or revise an existing one, to coordinate or oversee the work of river basin districts.

Serving flexibility and regional differentiation is positive, but, at the same time, the legal system must have "teeth" at the level of the EC if the Directive is to improve river basin management and be effective in the end. For chemical objectives, the key is to set enforceable standards, i.e. thresholds, for the most hazardous substances at the supranational level but allow for novelty and innovation in the manner in which Member States meet those standards. This raises the question of how the enforcement of standards not set at EU level can be made equally effective. The available oversight mechanisms of the monitoring and reporting of compliance with chemical standards are expected to achieve compliance with chemical standards set by Member States. This raises the question of whether the same approach is effective with novel ecological standards set by Member States. Guidelines for intercalibration are available to tackle the problems of comparability (EC, 2011a).

Compliance and enforcement in the US

The US EPA enforces requirements under the CWA, which was originally enacted in 1948[41]. The US EPA works with its federal, state and tribal regulatory partners through a comprehensive CWA compliance monitoring program to protect human health and the environment by ensuring that the regulated community obeys environmental laws/regulations through on-site visits by qualified inspectors and a review of the information the US EPA or a state/tribe requires to be submitted.

The website of the CWA[42] compliance assistance program provides businesses, federal facilities, local governments and tribes with tools to help meet environmental regulatory requirements. Under the CWA's National Pollutant Discharge Elimination System program, the US EPA regulates discharges of pollutants from municipal and industrial wastewater treatment plants, sewer collection systems and storm water discharges from industrial facilities and municipalities. The TMDL describes a value of the maximum amount of a pollutant that a body of water can receive while still meeting water quality standards (US EPA, 2015c). The Clean Water Action Plan targets enforcement for the most important water pollution problems.

Compliance, monitoring and sanctions

Table 3.4: Reasons of a regulate to respond positive or negative on regulation. Compliance behavior: “Table of 11”. Source: Van der Schraaf (2005).

Aspects of spontaneous compliance	1 Knowledge of the regulations 2 Cost/ benefit ratio 3 Degree of acceptance 4 Loyalty and obedience of the target group 5 Informal monitoring
Aspects of monitoring	6 Informal report probability 7 Monitoring probability 8 Detection probability 9 Selection
Aspects of sanctions	10 Chance of sanctions 11 Severity of sanctions

Legislation is the basis for the successful application of WQGs. However, the existing WQGs and related regulations show large differences in their approaches. They may be voluntary, market-based or mandatory, or combinations of these approaches. At a(n) (inter)national level, choices have to be made at to which subjects require mandatory approaches. Legal instruments may include rights and licences, taxes or charges,

[41] There were a large number of revisions and amendements, e.g. in the years 1956, 1961, 1965, 1966 etc. (Copeland, 2010).
[42] http://www.epa.gov/laws-regulations/summary-clean-water-act

penalties, but also the duty of monitoring and reporting. Besides the extent to which a rule is observed, the reasons for non-compliance will be examined. It is necessary to know the reasons because they will form the basis for selecting the appropriate intervention. Furthermore, the compliance behaviour of regulatees is the central point in all action that an authority takes to reach the policy goals identified (Van der Schaaf, 2005). The "Table of 11" is a methodology of identifying possible reasons for non-compliance. The Table of 11 was presented as an important part of a compliance strategy including spontaneous compliance, monitoring and sanctions. Sanctions are any adverse consequences imposed on a violator. Lugwisha et al. (2008) described the challenges of the compliance and enforcement of wastewater management legislation in Tanzania based on analysis with the Table of 11 (See »Table 3.4).

Stakeholder and public participation

Stakeholder participation and public participation is more and more recognized as one of the success factors for improving water quality and protecting ecosystems. It plays a crucial role in raising societal resilience and building adaptive capacity. In a number of WQGs and in basin organizations, participation is strongly advised or even a legal obligation. A few examples from different countries may illustrate that stakeholder and public participation is one of the leading principles in the improvement of water quality and the protection of ecosystems.

The NWQMS (NWQMS, 1998) in Australia stated that the national objectives will be achieved by applying four principles to water quality management, among which community involvement in setting water quality objectives and developing management plans. This policy has been applied in the development of Basin Plans. For example, the MDBA has presented a guide to the proposed Basin Plan in which the Authority is providing an early opportunity for individuals, stakeholders and the community to examine the thinking of the Authority and provide feedback. This feedback will be taken into consideration in finalizing the proposed Basin Plan. The Authority has developed comprehensive consultation and engagement processes. The steps the Authority must follow once the proposed Basin Plan has been released are outlined in the Water Act 2007. These include a minimum 16 weeks of public consultation providing individuals, stakeholders and the community with an opportunity to comment on the proposed Basin Plan.

The US guide to developing watershed plans to restore and protect our waters (US EPA, 2013d) provides six steps of watershed planning. It is important to note that the first step is “Build partnership”, including identification of stakeholders, identification of issues of concern, setting primary goals and undertaking public outreach. The US EPA has published a Public Participation Guide on the Internet[43]. It states that there is a great deal of public participation being implemented throughout the world today. Laws and regulations in many countries regularly require public meetings and comment on government actions. Some require even more extensive forms of public engagement and input. However, all of this activity does not automatically translate into good practice. Meaningful public participation requires much more than simply holding public meetings or hearings or collecting public comments. When done in a meaningful way, public participation will result in two significant benefits:

- Sponsor agencies will make better and more easily implementable decisions that reflect public interests and values, and are better understood by the public.
- Communities develop long-term capacity to solve and manage challenging social issues, often overcoming long-standing differences and misunderstandings.

[43] http://www.epa.gov/international-cooperation/public-participation-guide

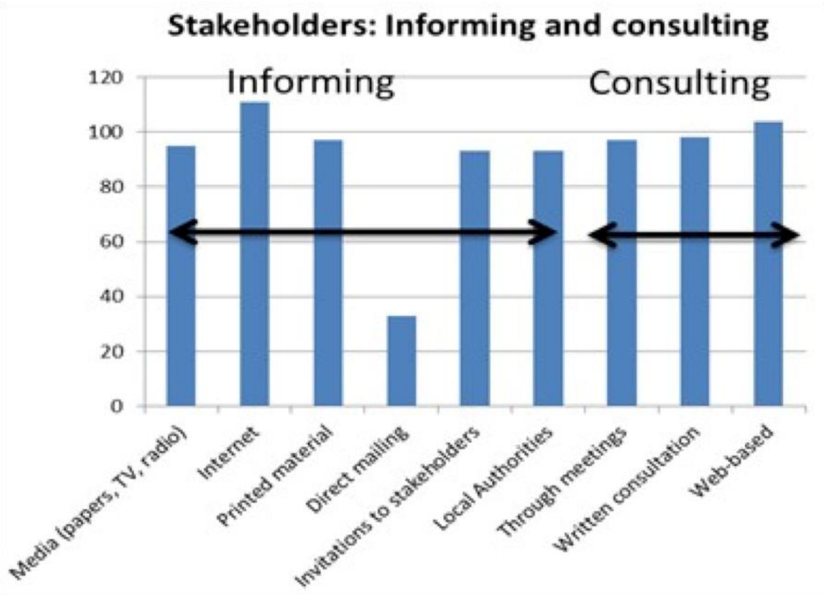
In the joint report of the WWF and Chinese experts (Pegram et al., 2013), ten golden rules of basin planning are given. One of these rules is “Engage stakeholders with a view to strengthening institutional relationships”. Basin planning should be seen as an opportunity to build trust and relationships between these bodies so that action to secure implementation can be achieved. The basin planning process should also recognize and try to incorporate the diverse perspectives of stakeholders at different levels that will have an influence on the implementation of the strategy.

The EU WFD sets out a framework for vertical coordination from the European level to the water body level as well as horizontal coordination of all relevant measures, stakeholders and policies requiring at least six months. The purpose is to involve all stakeholders, including the public, with a view to ensuring that the best and most cost-effective measures are identified and selected, and that acceptance of the measures is built into the process. Another key mechanism for sectoral and territorial integration is stakeholder involvement in the development of RBMPs by the requirement to 'encourage the active involvement of interested parties in the implementation' of the RBMPs, in particular in the development of plans, which sets out a three-stage process of stakeholder and public consultation requiring at least six months.

As public participation is considered as a key mechanism for integration and coordination at river basin district level, all RBMPs are evaluated regularly by the EC (EC, 2012). As shown in »Figure 3.11, the RBMPs indicated that a wide range of outreach methods and consultation mechanisms were used for reaching out to and consulting with stakeholders (including the public), (EC, 2012). The most predominant outreach methods were to use the Internet for announcing the consultation and carrying out the consultation by inviting comments via the Web. The media was used to a large extent for announcing the consultations and local authorities played a big role in reaching out. In many cases, the interested parties known to the authorities were directly invited to respond.

Figure 3.11: Means of informing stakeholders and the public, as well as consulting.

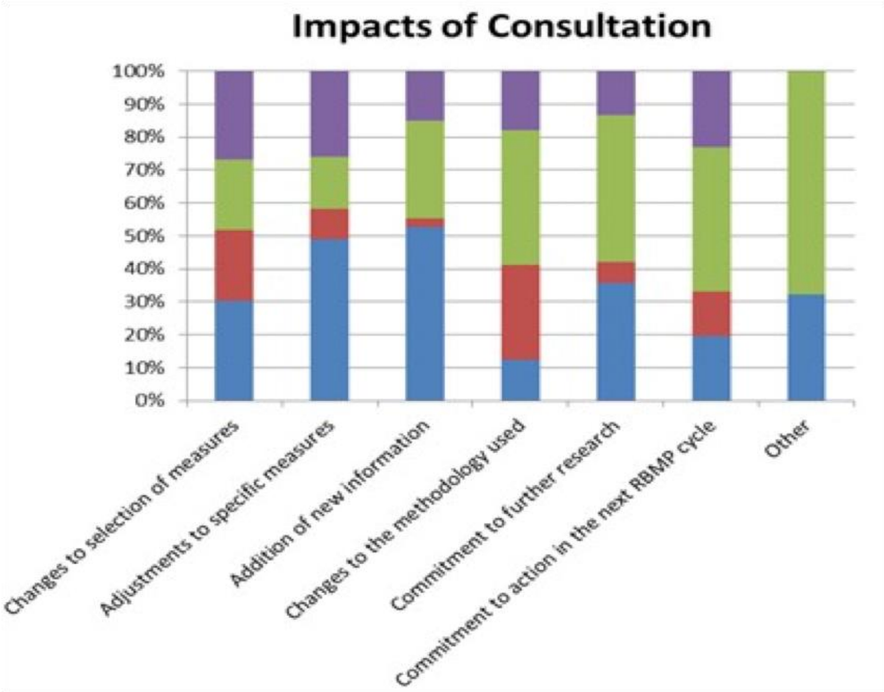
Source: EC (2012).



Although it is difficult to assess the real impact of consultations on the RBMPs due to the many responses by stakeholders, »Figure 3.12 gives an indication of the impacts on the main subjects of RBMPs. It appears that in some cases the consultation led to less stringent measures or objectives being defined, but in some cases an increased level of ambition was reported. In no other region or country has such a detailed analysis on the role of public participation been published.

Figure 3.12: Type of impacts of public consultation reported in the RBMPs.

Source: EC (2012).



Summary comments on application of water quality guidelines at basin level to protect freshwater ecosystems

The main findings concerning the application of WQGs at basin level to protect freshwater ecosystems are:

- The application of WQGs to protect fresh water ecosystems should be considered as part of Integrated Water Resources Management (IWRM). Given the fact that the basin approach has been widely accepted as the most suitable entity for IWRM, basin organizations may play an important role in the protection of ecosystems.
- In Australia, the EU and the US, the basin approach has been acknowledged as an entity and method which is important for the application of WQGs, but the role of basin organizations and the goals and impact of basin plans vary widely.
- The role of basin organizations highly depends on the type of basin organization and the mandate for the organization. There is no straightforward approach for the application of WQGs in basin organizations. Basin organizations can surely play an important role, but a tailor-made approach is needed in which the mandate of a basin organization and organizational structure should be established by the competent federal or national authorities, or in the case of international basins by the countries involved.
- Achieving compliance is a key element for implementing guidelines effectively. The need to achieve compliance by the public, companies and competent authorities is clearly recognized in programmes for implementation in the operational guidelines in Australia, the EU and the US. In addition to achieving compliance, legal and administrative means are indispensable to encourage or compel individuals to comply with their obligations under legislation.
- Basin organizations play an important role in overall water resource management and in achieving compliance, e.g. by encouraging stakeholders and public participation, and public reporting about the status of water quality.

- Impacts of public consultations are clearly demonstrated, e.g. in RBMPs in the EU (EC, 2012).

Conclusions

A quick review of WQGs in 15 countries or regions shows that in all countries and regions water laws and/or WQGs have been established to protect human uses and, in most cases, also to protect aquatic life. Most of the laws and guidelines date from the 1980s and 1990s and some have been partly adapted in recent years. However, only a few guidelines focus more explicitly on the protection of aquatic ecosystems by developing specific guidelines for this purpose. The Australian/New Zealand Water Guidelines, the EU WFD and related guidelines, and the US EPA guidelines were selected for a more in-depth review, as these guidelines are based on long-term experience and because they also most extensively provide science-based approaches and tools for the quality assessment of aquatic ecosystems.

The main conclusions based on the three guidelines reviewed:

- The term "Water Quality Guideline" has been used with at least two meanings:
  - i. the overall framework for assessment; and
  - ii. narrative or numerical criteria to assess water quality. Also, the terms WQC, water quality objective and WQSs should be defined clearly if WQGs are to be developed. The same holds for the terms uses, pressures and stressors, and also for thresholds and benchmarks.
- The guidelines for the aquatic ecosystem are part of a larger framework of guidelines for water quality, which may include guidelines for drinking water and other uses, analyses of pressures, pollution prevention measures, and monitoring and assessment methods.
- Quality classes for ecosystems are used for at least four reasons:
  - i. to formulate present or future objectives concerning the status desired;
  - ii. to present the ecosystem quality status in a transparent way;
  - iii. to create awareness by authorities and stakeholders; and
  - iv. to compare the quality status of different waters to report progress of the quality status.
- Narrative and numerical criteria for biological, naturally occurring physical and chemical quality indicators and hydromorphological indicators are nearly always related to RCs, e.g. the ratio of species observed for the site examined and known from reference sites.
- Comprehensive guidelines are available for deriving criteria for toxic substances. It is clear that deriving WQC is a complex process of the integration of high-level scientific knowledge, taking into account a large number of uncertainties and policy definitions of protection levels. The resulting numerical criteria in the guidelines considered sometimes show large differences mainly due to differences in the definition of the criteria level, the data used and safety factors applied.
- Frameworks and decision trees are available for the quality assessment of aquatic ecosystems. These frameworks and decision trees provide step-by-step approaches for quality assessment. Major elements are the setting of general objectives; typology of waters; methods for deriving quality criteria; biological, physical, chemical and hydromorphological indicators for monitoring and assessment; and methods for analysing and reporting monitoring data.
- Step-by-step approaches and decision trees are developed to support ecological assessment based on biological, physico-chemical and hydromorphological quality indicators as well as methods for the classification of the status of an aquatic ecosystem.

- The application of WQGs to protect freshwater ecosystems should be considered as part of Integrated Water Resources Management. The river basin has been acknowledged as an entity for IWRM and for the application of WQGs for ecosystems. However, in the countries and regions considered, the role of BOs and the goals and impact of basin plans vary widely.
- Compliance by stakeholders, companies and competent authorities is the key factor for the implementation of guidelines for aquatic ecosystems. However, enforcement mechanisms are indispensable for encouraging or compelling all stakeholders if they do not comply with obligations under legislation.

4 A Framework for Freshwater Ecosystem Management:

A framework centered on ecosystem health

Setting the context

The nexus of human well-being and freshwater ecosystem health

Human well-being depends on reliable access to freshwater of high quality and sufficient quantity, as well as the other natural resources and services healthy freshwater ecosystems provide (MEA 2005a,b; detailed discussion in »Section 2.3.1). Coherent guidelines are needed to fulfil the water quality-related objectives encapsulated in the various dimensions of human well-being. Utilitarian guidelines classify water bodies according to the degree to which they can appropriately serve as a basis for human use, such as agriculture, industry, domestic consumption, recreation, and fisheries. These kinds of guidelines and their supporting standards were established and subsequently embedded in national and international contexts during the 20th century (e.g. Ayers and Westcot, 1985; DeZuane 1997; WHO 2011).

Increasing pressures on water resources and the resulting stresses, along with a persistent lack of attention to the tight feedbacks between ecosystem functioning and human well-being, have led to an unprecedented deterioration of the hydromorphological, physico-chemical, and biological state (i.e. water quality in its broadest sense) of freshwater bodies globally. Indeed, it is becoming more and more evident that without adequate measures, such as continuously improving Waste Water Treatment (WWT) schemes, recycling water in closed loops, regulating Source-Directed Control (SDC) measures of pollution, and applying innovative and sustainable conservation strategies, many freshwater ecosystems still capable of supporting human well-being may collapse. Ultimately, it will cause long-lasting and likely irreversible consequences for human-wellbeing and the ecological integrity alike. Hence, there is an urgent need to establish Water Quality Guidelines (WQGs) that help to safeguard healthy and to restore degraded freshwater ecosystem. The key challenge is to manage freshwaters systems both– as a critical resource for humans as well as a highly diverse living entity.

In a world inhabited by more than seven billion people (reaching around some 10 billion by 2050) a humans-outside-nature approach will simply not work and can even be considered unethical. Without question, aspirations of human well-being (as expressed in the Millennium Development Goals (MDGs) and the Sustainable Development Goals (SDGs), as well as other multilateral agreements) need to be aligned with oftentimes competing environmental objectives. This dual role of meeting the needs of humans and nature has been identified in the SDGs for water (SDG 6), with parallel targets set for human wellbeing, pollution control and ecosystem protection.

The extent to which anthropocentric activities influence ecosystems and their capability for service provision can best be captured through a Drivers, Pressures, Stressors, States, Impacts, Responses (DPSSIR) based assessment (OECD 1993 and »Section 2.4 of this report). The driver(s), the aspiration of human well-being, exert inevitable pressures on the supporting ecosystems. As discussed in »Section 2.4, a distinction can be madebetween pressures related to human activities and (subsequent) stressors, which refer to the affected freshwater ecosystems (see also

Table 2.1 and »Figure 2.7). In fact, the level of pressures can be regulated to a great degree within the activities (e.g. selection of environment friendly green technologies) thus helping to mitigate or avoid triggering stresses or even threats for the freshwater ecosystem under consideration. Whether or not an ecosystem can cope with these pressures depends on the nature of the pressures, their magnitude, intensity, duration, frequency and interactions and whether they are superimposed to act as stressors.The state of the ecosystem co-determines its capacity to absor stressors without longlasting consequences. Healthy and diverse ecosystems are more resilient and therefore attenuate the effect of stressors.

In order to conceptualize the DPSSIR chain for freshwater ecosystems, the most relevant, direct stressors have been identified (»Table 4.1). The stressors then refer directly to the respective water body (freshwater ecosystem or part thereof), affecting it as point or non-point source influence. Stressors thus translate pressure(s) into specific and potentially negative consequences in or on water bodies. The status of a water body of concern can be characterized by

i.measuring the magnitude, spatial location and extent, and duration of stressors, by

ii.describing the state of the respective water body, and by

iii.quantifying the impact through the change of state, as a consequence of the stressor(s).

Stressors, state(s), and impacts can be measured (and monitored) by means of direct and indirect indicators. Indicators can be associated with a single or several stressors. They measure either the attributes of a stressor directly (e.g. phosphorus or nitrogen load) or characterize the state of the water body (e.g. its trophic state). Thus, indicators are either stressor- or state-related. In general, indicators can be grouped into three main categories, namely:

- hydromorphological indicators,
- physico-chemical indicators, and
- biological indicators,

with further subdivisions. Socio-economic indicators are not considered in the Framework at this time. They may provide useful hints and potential pressures and stressors, but these indicators are not monitored primarily for the sake of freshwater ecosystem health.

Table 4.1: Stressors of freshwater ecosystems and the respective types of direct indicators affected.

Stressors	**Indicators**		
	Physico-chemical	Biological	Hydromorphological
Water infrastructure	X	X	X
Flow alteration	X	X	X
Modification of aquatic habitat		X	X
Overexploitation		X	X
Biological water pollution (invasive species)		X	
Chemical water pollution	X	X	
Thermal water pollution	X	X	

The selection of indicators and the respective monitoring techniques employed depend largely on the water body type (although resource constraints also come into play). This volume considers three main water body types: lentic (lakes, reservoirs), lotic (rivers and streams), and palustrine wetland ecosystems (»Figure 2.2). . Flowing waters are further subdivided into permanent and temporary, and into wadeable and non-wadeable streams and rivers. The proposed stressors of freshwater ecosystems can primarily be characterized and monitored by a number of direct indicators, as listed in »Table 4.1. In case indirect indicators are applied, biological indicators are preferred because they reflect the alteration of freshwater habitats and biodiversity due to the aggregate effect of potentially several stressors.

Focus on freshwater ecosystems of inland surface waters

This volume focuses on freshwater ecosystems, which form the major part of inland waters. Freshwater ecosystems are the backbone of the entire water cycle, as they reflect and integrate the processes that occur in the surrounding terrestrial system, in the connected groundwater body, in estuaries and deltas, and ultimately also in coastal and marine waters.

Groundwater and surface waters are often intricately intertwined, for example, groundwater contributes to river base flow. At the same time groundwater bodies (aquifers) need to be recognized as ecosystems, while they provide 25 to 40% of the drinking water for humans globally. However, their biodiversity remains poorly known, with less than 10% of the estimated number of ground water species described so far. The number of described species in the surface water realm exceeds that of ‘groundwaters’ by 1-2 orders-of-magnitude (Balian et al. 2008, Stoch and Galassi 2010). While acknowledging the importance of groundwater as a major water resource, at this stage the Framework focuses on surface water bodies (lakes, reservoirs and running waters), as well as related transitional ecosystems, such as deltas, estuaries, and palustrine wetlands. Groundwater is dealt with implicitly. From the perspective of ambient water quality, for example, the quality of groundwater is manifested mainly in terms of natural outflows into surface water systems. Groundwater is also abstracted for utilitarian purposes and discharged into surface waters after use.

The concept of “ecosystem health” acknowledges the coexistence of human activities with ecosystem functions and dynamics. Ultimately, water bodies are recipients of the (waste) water flow and other residues. The incorporation of human activities seems relevant, especially when considering current human population density and demographic development. Furthermore, through the hydrological cycle even natural habitats at great distances from the source of human impact may be linked to intensively used or detrimentally impacted water bodies. In fact, this strong coupling between socio-economic and socio-ecological subsystems and the water cycle is much more pronounced for the surface water than for the subsurface compartments of the terrestrial part of this cycle. Consider the nitrogen cycle as an example. Nitrate enrichment in water bodies is often a consequence of intensive use of fertilizers in agriculture. In surface water bodies it may cause eutrophication and potentially toxic algal blooms, especially in lakes, which are clear signals that ecosystem health may be further and potentially irrevocably compromised. Consequently, immediate measures are required, preferably at source. Nitrate levels can increase in groundwater bodies too with, for instance, percolation from agricultural areas ultimately rendering groundwater unfit for human consumption. Both the occurrence of the problem and its reversal in groundwater bodies are slower processes, however, than in surface waters. Remedial actions preventing the deterioration of surface waters are also beneficial to mitigate the threat for groundwater bodies, as well as the coastal and marine environment.

A focus on freshwater ecosystem health

The Governing Council/Environment Assembly (GC/EA) of the United Nations Environment Programme (UNEP) Decision 27/3 (»Annex 1) clearly refers to “water management” and to “water quality”, which implies the focus on the aqueous phase of ecosystems (see also »Section 1.1). As water use-oriented guidelines (drinking water, recreation, fisheries, livestock, agriculture, etc.) already exist, the specific focus of the Framework is on freshwater ecosystem health (see also »Box 4.1).

Good ecosystem health supports services, although overusing these services may in turn result in a deterioration of ecosystem health. For example, it could be argued that if a lake has a good fishery it is healthy, even though that fishery is a result of increased eutrophication, which concurrently may result in the demise of other biotic components - such as the loss of macrophytes or the switch to a turbid state. The introduction of Nile Perch into Lake Victoria, East Africa, is another example. The lake supports a large fishery as a result (although now overfished), but has at the same time experienced the decline or total elimination of its native and endemic cichlids. Hence, ecosystem health assessment requires a complementary set of indicators to avoid biased outcomes and decisions based on insufficient scientific evidence.

Box 4.1: Overuse of ecosystem services

Applying an ecosystem health approach in the Framework necessitates adopting the precautionary principle (»Section 2.5.2). It acknowledges that freshwater ecosystems have a legitimate demand for an appropriate part of the resource, enabling them to sustain their fundamental functions and related services. Only healthy water bodies can provide and secure their respective Ecosystem Services (ES) in a sustainable way. The Framework adopts the precautionary and the no-further-deterioration principles. This is in contrast to the prevailing “impair-and-then-repair” paradigm , which is still common practice in water resources engineering and development (Vörösmarty 2013, see also »Sections 2.3.2 and 2.3.3).

Water quantity and thus the link to hydrological regime alteration (i.e. changes in water flows and/or levels), and its mitigation through environmental flow management, is another major component of ecosystem health. Changes in water quantity and the degradation in water quality jointly threaten the structure, functioning and resilience of freshwater ecosystems. Water quantity, its adequate spatial availability, temporal distribution, and the geomorphology and physical habitat template it maintains, control biodiversity and ecosystem processes and services. Furthermore, the maintenance of environmental flows as an essential, complementary management approach alongside the establishment of WQGs and standards is emphasised as critical for success (»Section 2.3.3 and »Box 2.2).

The hydrological regime, and its variability and predictability within and among years, can be characterised in terms of the following flow components: extreme low flows, low flows, high flow pulses, and small and large floods. Each component or type of flow event can be further described in terms of the criteria of magnitude, timing, duration, frequency and rate of change. Water flow can be thought of as a master variable that directly and/or indirectly influences the main ecological structural and functional attributes of streams such as: water quality, physical habitat, energy supply, biological composition and species interactions, and connectivity in space and time. In fact, the practice of setting environmental flows (e.g. see Tharme 2003; Bunn and Arthington 2002; Poff et al. 2010; Arthington 2012) is underpinned by the natural flow paradigm, which recognises that “The full range of natural intra- and inter-annual variation in hydrologic regimes, and associated characteristics of timing, duration, frequency, and rate of change, are critical in sustaining the full native biodiversity and integrity of aquatic ecosystems.” (Poff et al. 1997).

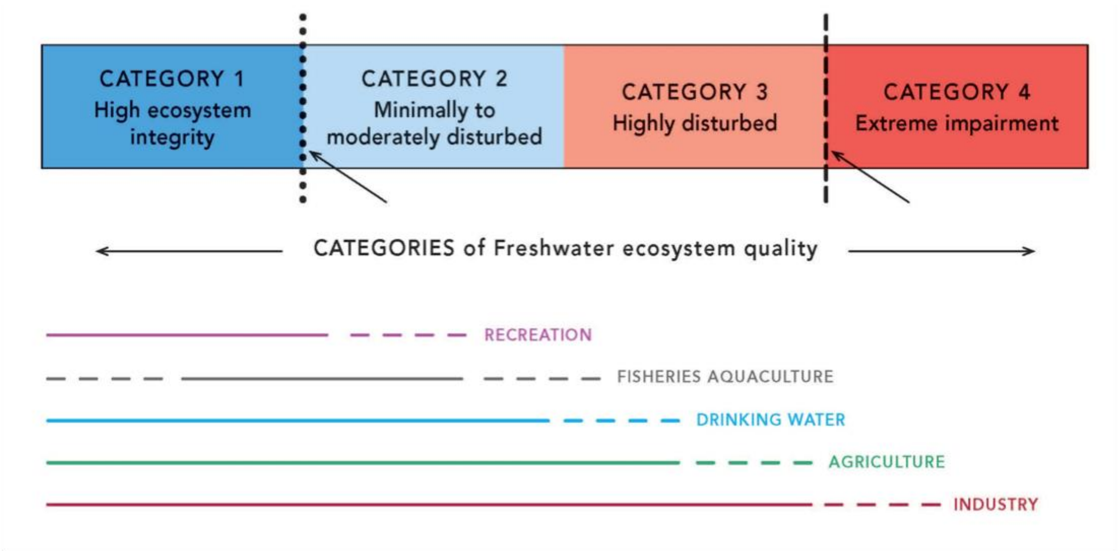
Water quantity and water quality are inextricably interrelated, and fundamental in sustaining freshwater ecosystem health and the livelihoods and well-being of dependent communities. There is strong scientific evidence that alterations of the flow regime can affect water quantity, quality and changes biodiversity. For example, fish species richness can significantly decrease with decreasing flow magnitude, native riparian vegetation depends on natural water level fluctuations for its vigour and structural composition, and specific high flow events, such as large floods, are required for the formation of key habitats along river corridors (e.g. sand and cobble bars and islands). Aquatic biotas are adapted to and dependent on these natural spatiotemporal variations in water quality tied to river flow regimes (Bunn and Arthington, 2002; Nilsson and Renöfält, 2008). Flow-related changes in chemistry, often complex and difficult to predict quantitatively, can exert a wide range of potential stresses on riverine biota which may become pronounced under extreme flow conditions, such as very low or drought flows (Nilsson and Renöfält, 2008).

Classification of the health of freshwater ecosystems

The Framework considers four principal water quality categories arranged along a quality continuum. These categories are associated with different states of ecosystem integrity and health of the respective freshwater system. A first benchmark (»Figure 4.1, dotted line) separates ecosystems of high integrity from all other categories – ecosystems in this category include highly intact or near-natural systems, often associated with areas of high conservation value (e.g. protected areas such as national parks, Ramsar wetland sites, or Key Biodiversity Areas (KBAs)). A second benchmark (»Figure 4.1, dashed line) demarcates the lower end of the quality continuum, a category where water quality is deteriorated in such a way that it does no longer sustain higher multicellular life forms. While water bodies below the "dashed line" benchmark may exist in rare cases naturally (e.g. hot springs, hypersaline waters, vulcanic lakes), they are usually not able to provide a broad range of ES for human well-being.

Figure 4.1: Categories of ecosystem status along the freshwater ecosystem quality continuum and different potential utilizations of water along the categories of freshwater ecosystem quality (coloured lines are illustrative of different water uses: continuous lines indicate "water fully usable for respective purpose", dashed lines indicate " water quality just of sufficient quality for respective purpose", no line indicates "water quality not sufficient for respective use").

Note: within agricultural and industrial use, variability in the quality of source water needed can vary depending on the crop and industrial process, respectively.



Different water quality categories may be associated with predominant ES and human uses. As »Figure 4.1 illustrates, ecosystem functions and withdrawal or in-stream/in situ use of water for various uses (as illustrated by coloured lines) may be compromised with deteriorating water quality.

With its proposed four categories, the Framework provides a state-of-the-art approach as a basis for further refinement, if needed. (for comparison: different classifications used in different countries (see »Chapter 3). In the following, the proposed freshwater ecosystem health categories are explained in more detail.

Governance underpins all aspects of the four Phases of the Framework.[24] There are many frameworks for ecosystem and water governance.[25] One of the most established is the Integrated Water Resources Management (IWRM) approach, which is also monitored through SDG indicator 6.5.1. The IWRM approach is suitable for freshwater ecosystem governance.

Governance may be broken down into the following four components[26]:

1. Enabling Environment
2. Institutions and Participation
3. Management Instruments
4. Financing

Category 1 (high ecosystem integrity)
Within this category ecosystems provide the whole set of goods and services at high level, qualitatively and quantitatively. Irrespective of any direct human use, the ecosystems fulfil their key functions and processes such as nutrient cycling, decomposition, or the unimpeded dispersal of biota. The systems have the capacity to recover rapidly from natural disturbances, i.e. they exhibit a high degree of resilience. This near-natural or intact state corresponds with the so called "reference state" as defined and used in a number of existing guidelines. Ecosystems falling under Category 1 can be identified as areas of conservation planning priority.

Category 2 (minimally to moderately disturbed)
This category represents freshwater ecosystems which are minimally to moderately disturbed. The quality status of this category would allow direct human contact through swimming, recreation, and water withdrawal for drinking purposes with minimal treatment needs. Indirect human water uses, like irrigation withdrawal, artificial recharge, absorbing well treated sewage, water transportation, and hydropower use can be accepted, though these may already imply hydromorphological and other stresses. In this category there is a clear trade-off between ecosystem integrity/functions and ecosystem service uses. The full functionality of the ecosystems might be affected. This category is expected to provide habitat for sensitive species of fishes like the salmonid to cyprinid group in temperate and cold climates and equally sensitive species as indicator taxa in subtropical and tropical water bodies. A threshold of concern could be set to identify the boundary condition within this category and trigger an appropriate management response to reverse degradation and ensure that the referent water bodies remain in acceptable and move towards good ecological condition.

Category 3 (highly disturbed)
This category represents highly disturbed ecosystems and entering this threshold of concern (dashed line in »Figure 4.1) should trigger immediate management action to identify and address the source(s) of the problem and avoid unacceptable degradation. There is a clear negative impact of land and water use on ecosystem integrity/functions. ES may no longer be available for the purpose of drinking water supply, recreation or fisheries and aquaculture, whereas freshwater quality may be sufficient for insensitive industrial utilizations, navigation and potentially hydropower generation.
This represents a status of the ecosystem below a threshold value(s) requiring urgent remedial action to restore the ecological functions of the respective water body. The resilience of these ecosystems is low and the status is possibly at a point of no return without massive remedial actions which ultimately may result, even if successful, in purely artificial, man-made ecosystems.

Category 4 (extreme impairment)
An ecosystem health bottom line (dashed line in »Figure 4.1) is defined. Ecosystem quality status should be above this threshold. Below this threshold most aquatic diversity and beneficial use is limited and ecosystems will face severe reduction or loss of ES. It is acknowledged that water quality conditions corresponding with characteristics of this category may occur naturally due to volcanic activities or other geogene processes. For such water bodies, ecosystem health is unlikely to be a useful concept but they also do not support human well-being. This category represents an unacceptable status of ecosystems whereby the status is characterized by threshold values indicating the urgent need of remedial action to restore the ecological functions of the respective water body. The naturalness of these ecosystems is low and the status is possibly at a point of no return without massive remedial actions resulting ultimately in man-made ecosystems.

It is acknowledged that water quality conditions corresponding with characteristics of the 'Category 4 domain' may occur naturally due to volcanic activities or other geogene processes. For such water bodies ecosystem health is unlikely to be a useful concept.

The proposed Framework for developing water quality guidelines for freshwater ecosystems

N.B. The proposed Framework described below differs slightly to the revised Framework presented in Volumes 1 and 2 of this series. See Preface of this Volume for details.

The aim of the Framework, as derived from operational paragraph 1 of the UNEP Governing Council Decision 27/3, is to develop a framework of science-based policy and technical recommendations for policy and practice. It is developed to accommodate the broad context of climatic, biogeographic, and hydromorphological conditions of individual freshwater ecosystems. These characterizations introduced in the Framework should be all-encompassing, yet not overly complex or unnecessarily detailed in their characterization in order to facilitate its wide application. It should be kept in mind that every standard, parameter, and monitoring requirement may have serious and long lasting costs and capacity implications. Professional and institutional engagement and their respective budgetary support are likely to be present in the decades-long cycles of adaptive water resources management. The Framework therefore offers modular, stepwise options for indicators and monitoring practices, thereby enabling an adaptive implementation of assessment and remediation schemes. The adaptive water quality assessment and management approach of the Framework (see »Figure 4.2) is composed of four main phases:

I. Initiation Phase: Definition of the study area for which the guidelines will be used and setting of objectives covering Steps 1 – 3.
II. Identification Phase: Collecting and optimizing the use of existing data, knowledge and information covering Step 4.
III. Assessment Phase: Collection of new data and assessment of prevailing conditions and trends, framework setting for classification, selection of indicators, monitoring and evaluation of the data including reporting covering Steps 5 - 7.
IV. Policy Development Phase: Integrating the guidelines into adaptive management and governance frames and setting priorities for further assessment and management of freshwater ecosystems covering Steps 8 – 9.

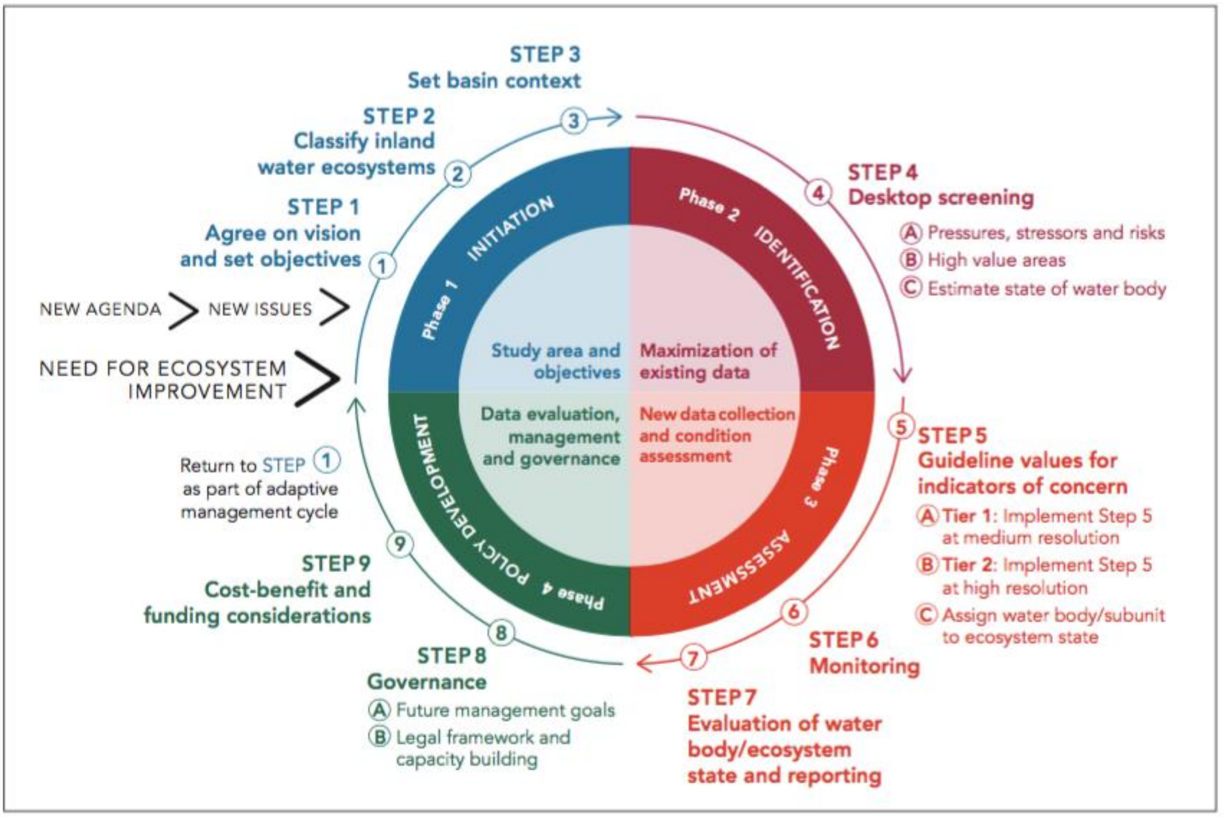
These four phases are subdivided into nine steps (in some cases with further subdivisions). In addition, the whole procedure is conceived as an adaptive management loop with a regular feedback, potential modification, and subsequent re-run of Steps 1 – 9. Steps 6 and 7 have to be revisited in short-term (annual) cycles. The entire cycle (Steps 1-9) needs to be revisited in 5-10 years. Unscheduled reviews may be required by unexpected events, development and/or deterioration of freshwater ecosystems.
Periodic checking and revision of the guidelines and their phases is an essential component of their use. The relevant time frame relates to the capacity to monitor and the institutional framework within which the guidelines are used. It is recommended to review these components once in a ten-year cycle. »Figure 4.2 shows the nine steps within the four phases.

Beyond this recommended decadal review of the 4 phases – 9 steps smaller iterative cycles (like revising ecosystem health class assignments in Step 5c should the evaluation and reporting of monitoring results in Step 7 prove their infeasibility) should be envisaged and implemented more frequently.

Moving from Tier1 towards Tier2 approaches (as shown in »Figure 4.2) the diagnostic capability and confidence in the conclusions increase. The decrease of uncertainty, however, is associated with higher costs and more sophisticated technical requirements. There are obvious trade-offs among these factors.

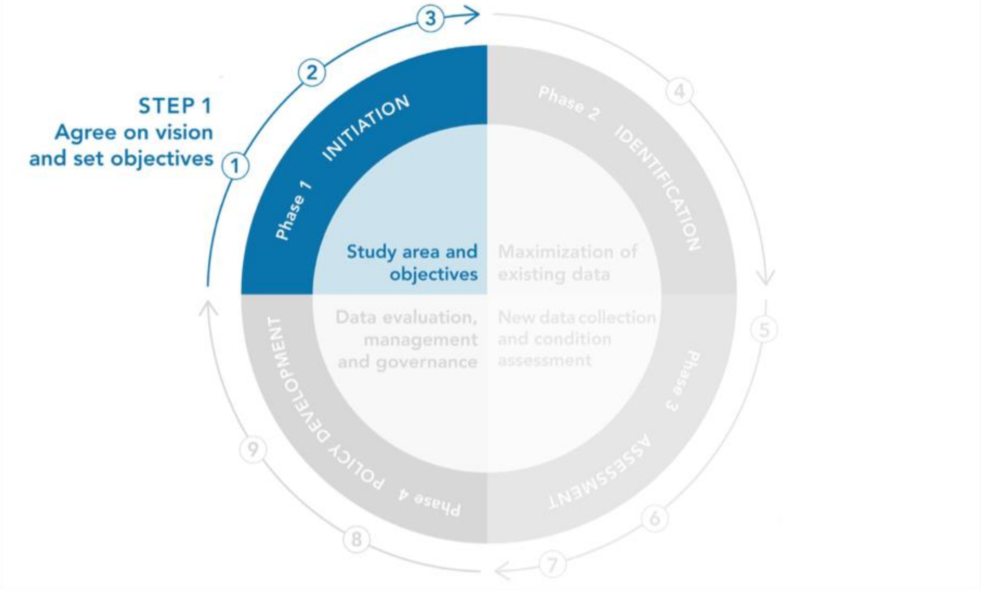
A more detailed version of the adaptive water quality assessment and management approach of the Framework (as shown in »Figure 4.2) is provided in »Annex 6.

Figure 4.2: Overview of the 4 phases 9 steps approach in the Framework.[44]



Setting objectives (Step 1)

N.B. This step matches the 'Set vision and objectives' step in the revised Framework (See Preface).



From vision to objectives

WQGs need to be rooted in an agreed vision on desired ecosystem use and respective water quality objectives. Focusing on WQGs for ecosystems implies a societal agreement on

[44] The proposed Framework described in this Chapter differs slightly to the revised Framework presented in Volumes 1 and 2 of this series. See Preface of this Volume for details.

considering - next to human uses - the intrinsic value of ecosystems including their biodiversity and related goods and services. This relates to wider perspectives and goals for sustainability as agreed by the UN SDGs[45], not only for the specific water goal (SDG 6 and targets), but their connection with the other 16 SDGs. Water quality objectives are set to protect ecological values and uses, as well reducing pressures and stressors, preventing further deterioration of water bodies, and thereby achieving improved status of freshwater ecosystems towards a defined reference state representing a "Least Disturbed Condition" (LDC) or less ambitious (and in many circumstances more realistic) "Best Attainable Condition" (BAC). Promotion of sustainable water uses and ES needs to be accompanied by regulations that ensure monitoring the ecological state of water bodies and that progress reporting, stakeholder involvement and enforcement of quality standards. Water quality objectives are preferably the result of informed decision making based on a societal discourse involving a large variety of stakeholders, many of whom may have potentially conflicting interests. In order to identify which stakeholders are involved a stakeholder analysis has to be set up (Schmeer, 1999). Questions like 'What does a 'healthy' river mean?' need to be effectively discussed. Such questions relate to what does society expect either to be able to catch more fish in the future, or maintain or even increase fish species diversity? Are there specific iconic species or species of conservation significance that need to be protected? Unless specific objectives and values are identified and agreed, it is difficult to determine the specific indicators that should be applied, and set the appropriate benchmarks or thresholds. Furthermore, it remains a challenge to justify public investment in monitoring and even harder to argue for management interventions to protect or restore ecosystems when only problems are identified but objectives not agreed (Bunn et al. 2010).

Objectives are distinct to WQGs but ideally they are tightly linked in two ways:
i.existing technical WQGs might be used as an important input to the development of water quality objectives because they can be used as a starting point, and
ii.a set of (new) objectives may support the development or adjustment of existing guidelines, also as part of an adaptive cycle.

An adaptive cycle is especially relevant because the establishment of a vision and objectives requires stakeholder inputs as well as the consideration of constraints posed by social and economic pressures, potential stressors and their impacts.

Objectives can contribute to the vision directly (i.e. fundamental objectives) or indirectly and can exhibit a hierarchy according to factors such as the certainty that they can be measured, or less well-defined but more normative descriptors. Hierarchical objective setting might facilitate and stimulate explicit discussion among stakeholders. An example for an objective hierarchy would be: rehabilitated river section > high level of ecological integrity > high level of chemical integrity > low pollutant concentrations (Reichert et al. 2007).

Set well-defined objectives

Objectives need to be well defined and their achievement preferably measurable to allow assessment of the degree of achievement. Hierarchical objective setting supports the identification of measurable indicators. Reichert et al. (2007) provide objectives and indicators for achieving physical, chemical, biological and socio-economic integrity for river systems. For example, 'high level of chemical integrity' can be assessed with the 'level of pollutants' which could be characterized e.g. by the measurable indicators 'mean metal concentration' or 'mean organic pollutant concentration' (Reichert et al. 2007). These indicators are often quantitative and their agreed numerical values can serve, as used also in different guidelines referred to as reference values, criteria or standards. They are used to differentiate between water quality states from very poor system states (dead zones, highly polluted/ high disturbance) to natural or near-natural ecosystems with high ecological values. Concerning physical/chemical indicators, a large number of quantitative water quality indicators (and their suggested values as thresholds or benchmarks) are available. Indicators for biological water quality objectives may be less easy to establish than for physical/chemical ones, because of inherent variability

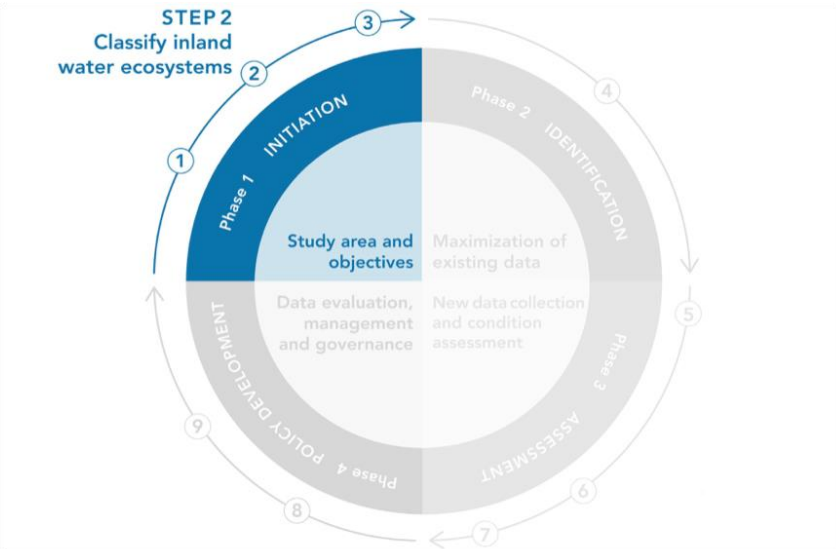
[45] United Nations (2015): Sustainable Development Goals, https://sustainabledevelopment.un.org/?menu=1300

of biological communities. Both biological and chemical benchmarking can, further, be strongly influenced by the geographical characteristics of the water body. Often reference states are used to compare and describe the biological attributes of a water body (also using normative or qualitative descriptors). Indicators for hydromorphological objectives refer to the hydrological characteristics, the morphology and the hydraulics of a water body.

In summary, it is important to identify the biodiversity and ecosystem values consistent with an agreed vision, and the water quality objectives that/ecosystem health will protect those values. Without a clear vision and measurable objectives, a monitoring programme will be poorly defined and unlikely to lead to significant improvements in ecosystem health.

Typology and classification of the health of freshwater ecosystems (Step 2)

N.B. This step is split into the 'Design classification frameworks' and 'Identify ecosystems and classify by type' steps in the revised Framework (See Preface).



As noted in »Section 2.2 some form of classification of water body types is required when developing a monitoring programme to acknowledge their natural differences, because:

- different types of inland water bodies will not look and function the same way even when they are healthy;
- the types of indicators that might be appropriate in one type of water body may not be appropriate for another;
- the methods used to sample one type may not be applicable or relevant in another; and
- even when the same indicator can be used in different inland water body types, the threshold or target values are likely to differ (Karr 1999; Bunn et al. 2010; Hering et al. 2010)

It is important to note that if a framework for typology of water body types already exists for a particular country or basin, the existing framework should be fully considered and, if possible, used to assure continuity with existing programs and policies that are in place. Many countries have national wetland inventories, which are an important source of information on the types, spatial extent and locations, and status of water bodies.
A simple typology scheme is described in »Section 2.2 including a proposed classification of inland waters (»Figure 2.2). It considers three categories of freshwater bodies (ecosystems):

- Running waters (permanent or temporary and wadeable or non-wadeable streams and rivers, riverine wetlands, riparian floodplains)

- Standing water (canals, ponds, lakes, reservoirs, lacustrine wetlands)
- Palustrine wetlands (e.g. fens, swamps, groundwater-dependent wetlands)

This typology is valid for both natural and artificial water bodies. There is a need to make the distinction between temporary and permanent water bodies in different climatic zones, where the phenomenon of drying occurs naturally, and healthy ecosystems are able to cope with intermittency (Acuna et al., 2014). Running waters (water courses) can also be subdivided as wadeable and non-wadeable streams and rivers to account for the differences in applicable monitoring techniques.

The three categories of freshwater bodies need to be considered under different climatic factors:

- Temperature: cold, temperate (moderate), sub-tropical, tropical
- Hydrology/precipitation: arid (dry), semi-arid, sub-humid, humid (wet)

Finally, and especially for streams due to their profound hydromorphological changes, but also for lakes due to their different stratification patterns, the following three zones have to be considered:

- Upland/Mountain
- Mid-reach/Mid-level (piedmont or plain)
- Low land (including delta, estuary and coastal zones)

These three zones may correspond to different stream orders (1-3, 4-6, >6) and/or geomorphic river types (constrained, braided/anastomosing, meandering). Stream order and/or river type are linked to specific biota and characteristic ecosystem processes and may therefore be used as proxies to assess the ecosystem state (e.g. braiding index or sinuosity as geomorphic indicators; production to respiration ratio as an indicator for ecosystem processes).

See »Section 2.2 for further information on the process of the establishment of a classification of inland water ecosystem types as well as an overview of typologies and already adopted classifications.

As far as classifying the ecosystem health status of freshwater bodies is concerned, the Framework promotes the use of a multiple category system like the one described in some detail in »Section 4.1.4. More elaborate classifications (using more than four categories) can be envisaged. However these more elaborate schemes mean an improvement of classification only if they are supported with a good quality detailed data base and sustained, quality controlled monitoring programme.

Setting the basin-scale environmental context (Step 3)

N.B. This step matches the ‘Set basin context’ step in the revised Framework (See Preface).

Delineation of catchments

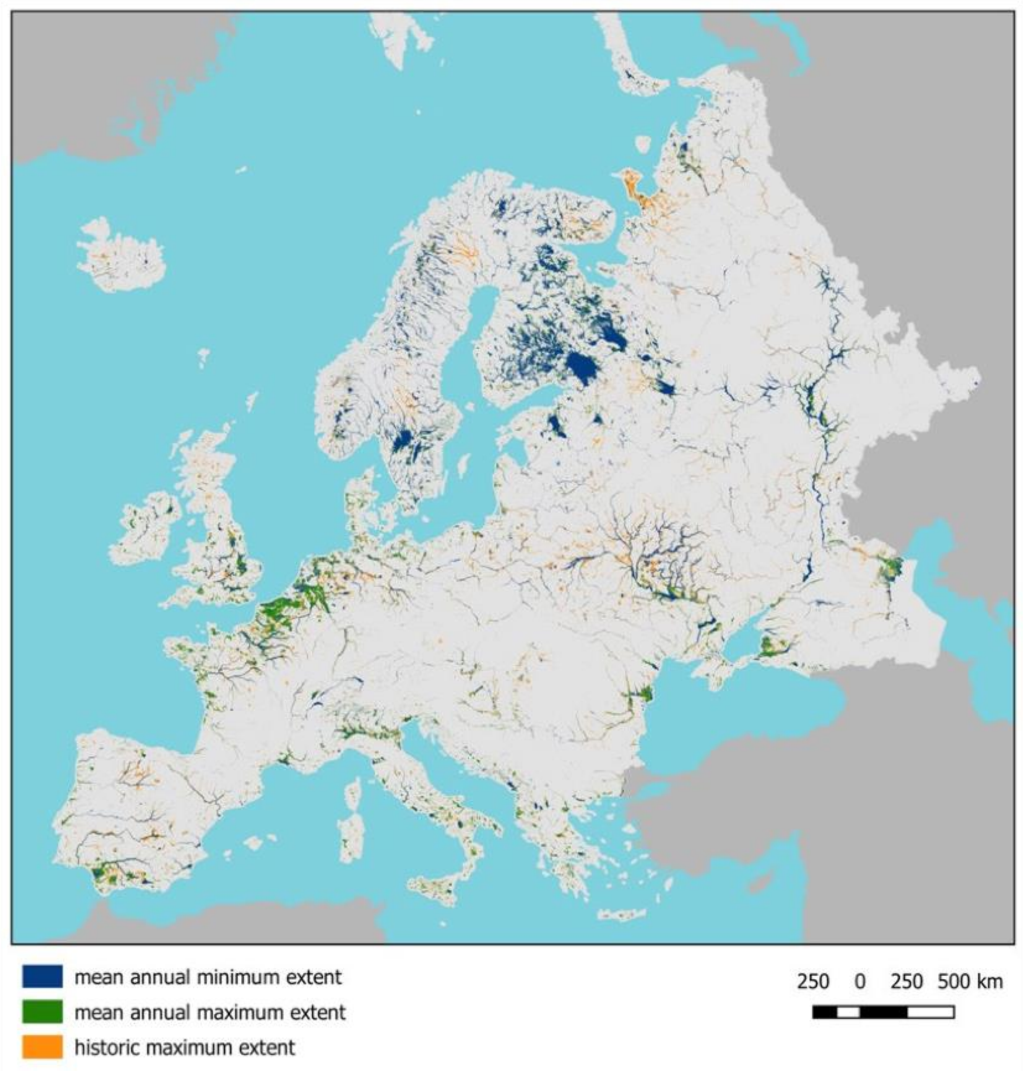
The catchment (river) basin, also called the watershed, is the core unit for freshwater ecosystem assessment and management. For obtaining accurate hydrographic information for catchments at the global scale and in a consistent format large geo-referenced data sets are available. HydroBASINS is a global river and lake catchment layer, derived from HydroSHEDS[46] (Lehner, 2012), and the global lakes and wetlands database (GLWD)[47], provide the most accurate hydrographic information for catchments at the global scale and in a consistent format (http://www.freshwaterplatform.eu/). It offers a suite of geo-referenced data sets (vector and raster) at various scales, including river networks, river and lake catchment boundaries, drainage directions, and flow accumulations. HydroBASINS is based on high-resolution elevation data obtained during a Space Shuttle flight for NASA's Shuttle Radar Topography Mission. HydroBASINS can be used as an open-source, standardized, basic global information layer for assessing the ecological state of freshwaters. River and lake

[46] www.hydrosheds.org http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database
[47] http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database

sub-catchments within HydroBASINS form the spatial unit for mapping and assessing species distributions and associated environmental parameters. HydroBASINS information is available at 12 spatial resolutions (corresponding to different watershed delineations according to the so-called Pfafstetter levels), which allows a nested analysis of pressures, stressors and response variables, depending on objective definition and the availability of data (e.g. regional effects on local conditions) (Lehner 2012, Fluet-Chouinard et al. 2015). Furthermore, a global river network map, a global freshwater wetland map, and global information on the distribution of lakes and reservoirs are already available or under development (Example of Global Inundation Map: »Figure 4.3).

The Global Lakes and Wetlands Database is another important geo-reference data set[48]. The combination of best available sources for lakes and wetlands on a global scale, and the application of Geographic Information System (GIS) functionality enabled the generation of a database which focuses in three coordinated levels on (1) large lakes and reservoirs, (2) smaller water bodies, and (3) wetlands.

Figure 4.3: Inundated areas of continental Europe derived from the Global Inundation Map. The map shows estimates for average annual minimum extent (representing the dry season extent), average annual maximum extent (representing the wet season extent), and long-term maximum inundation at 15 arc-second spatial resolution. Coastal inundated areas in a distance of 5 kilometers from the coastline are not included. Source: Fluet-Chouinard et al. (2015).



Biophysical context of a catchment

Baseline information on the general characterization of the catchment can be derived from publicly available databases. It is a desktop study that collates and combines the key

[48] http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database

background information on pressures/stressors and response variables. A first set of catchment-specific indicators that characterize the state of the main pressures (e.g. land use, degree of fragmentation, human population density, water abstraction, atmospheric nitrogen deposition), and the deviation from no/low pressure conditions, can be calculated based on existing background information. If available, these indicators should be cross-validated using ground-based data or multiple data sources. Furthermore, simple modelling approaches (e.g. regression analyses) should be applied to forecast trends under various development scenarios (e.g. climate change, human population development, economic development), at least to assess the expected direction (increase, decrease), degree (low, mean, high), and probability of change (see, for example, approach applied in the IPCC report).

Required baseline information – setting the context

- In the following the key issues and basic information are itemized:
- Biogeographic setting (Ecoregions of the world; http://www.feow.org/)
- Hydrometeorologic setting (precipitation and temperature patterns, specific runoff)
- Topographic setting (relief, geology)
- Environmental setting (e.g., background information on soil conditions, baseline deposition of nitrogen and phosphorous)
- Demographic setting (human population density and development, economic status and trends, e.g. Gross Domestic Product (GDP))
- Land-use setting (relative proportion of standardized land-use categories and rate of change in % per time unit)
- Freshwater ecosystem setting (freshwater ecosystem types, river network density, lake area and density, wetland types and area) and calculation of a freshwater ecosystem diversity index (see »Section 4.4.3 and Box 4.2 for details), which integrates composition, spatial configuration, and connectivity of freshwater ecosystem types
- Freshwater biodiversity setting (predicted/expected species diversity, KBA based on the proportion of unique and threatened species)
- Conservation setting (% and distribution of protected areas of different categories such as national parks, NATURA 2000 areas, Ramsar sites)

A freshwater ecosystem diversity index would be a major step forward in assessing river systems. In particular because it would link biodiversity with ecosystem services and would focus on ecosystem diversity; the latter being an often neglected component of biodiversity.
Ecosystem diversity and/or complexity: A landscape-based index that considers the composition, configuration and connectivity of freshwater ecosystem types can be calculated for each catchment unit (i.e. different spatial scales). This index can be compared to the historic ecosystem diversity based on reference/historic material such as maps and air photos. Development and application of such an index, or the adaption of existing landscape diversity indices (e.g. Soininen et al., 2015) for the required purpose would, as a first step, need to include and calculate the number of ecosystem types, the relative proportion and the evenness of ecosystem types, and the associated functional properties of ecosystem types; in line to the assessment

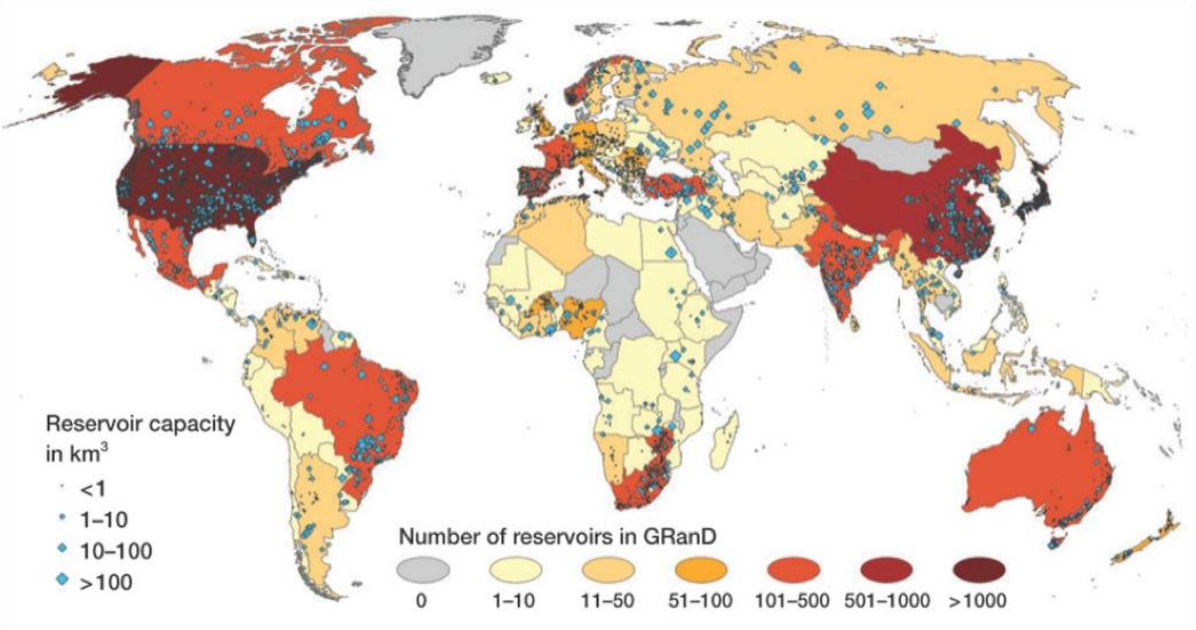
Box 4.2: Freshwater ecosystem diversity

It is proposed that thematic maps, based on the available data bases that are in most cases publicly available, while accepting intellectual property rights, are developed. A set of open access global or continental data bases can be used such as land-use (e.g., Corine land cover types[49]) or biodiversity data (e.g., Global Freshwater Biodiversity Atlas[50]). Two examples: first, the hydropower data base allows assessing the degree of fragmentation of the main river basins (Grill et al., 2015). It is based on the global Reservoir and Dam database (GRanD),

[49] http://www.eea.europa.eu/data-and-maps/data/corine-land-cover-2006-raster-3
[50] http://atlas.freshwaterbiodiversity.eu/

which, as of early 2011, contains information regarding 6862 dams and their associated reservoirs, with a total storage capacity of 6197 km3. On the basis of these records, it is estimated that about 16.7 million reservoirs larger than 0.01 ha exist worldwide, increasing Earth's terrestrial surface water area by more than 305,000 km2 (Lehner et al. 2011, »Figure 4.4). Second, the International Commission for the Protection of the Danube River basin (http://icpdr.org), for example, uses such thematic maps as a decision tool for conservation and management planning at the (sub-) catchment scale (development of River Basin Plan, Implementation of Integrated Water Resources Management (IWRM)).

Figure 4.4: Spatial distribution of existing hydropower dams worldwide according to the GRanD database.
Source: http://www.gwsp.org/products/grand-database.html and Lehner et al. (2011).



Desktop screening (Step 4)

N.B. This step matches the 'Desktop screening and assessment' step in the revised Framework (See Preface).

The identification phase, step 4, is a screening at desktop level for pressures and stressors defining the state of water bodies and verified information by stakeholders. This step includes three sub-steps:

i. Screening of pressures, stressors and risks,
ii. Identification of high-value areas and water bodies for protection, and
iii. A first estimate of the present state of each water body.

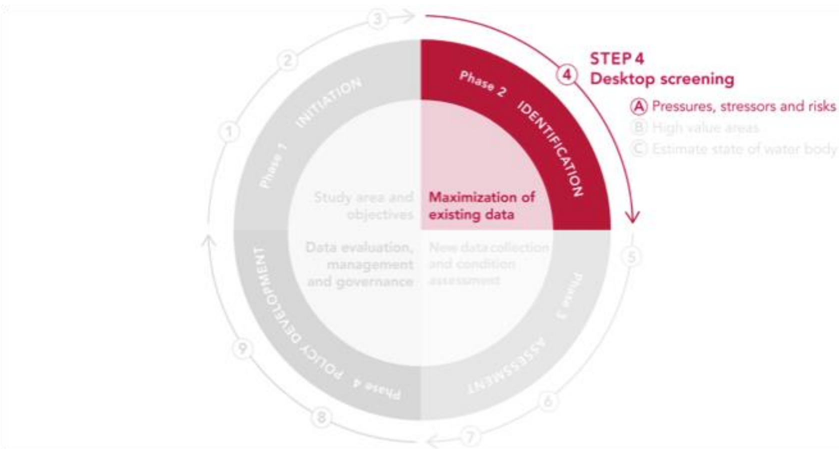
General principles applied in the Framework for assessment

- The catchment (watershed) is the basic unit for water management, therefore also for freshwater ecosystem health assessment. In many countries, however, healthy aquatic ecosystems are characterized by the length of rivers in the respective category. Though this does not exclude that different water bodies or parts thereof could have different designated uses and/or aspirational goals (e.g. using the HydroBasin/HydroShed layers at global scale; http://hydrosheds.org/)
- Reference conditions (RCs) are defined as near-natural conditions that form one endpoint (benchmark) along an ecosystem health continuum. The near natural conditions are defined based on indicators that reflect pressure/stressor values (Step 4a) as well as on indicators that reflect the physico-chemical, biological and

hydromorphological conditions of a specific ecosystem type (Step 4c)

- Best attainable conditions are primarily defined as a management option/objective. This may be close to the RC, but this implies frequently that a less than undisturbed level of ecosystem health is settled as an aspirational target that can be achieved with best management practice.
- Develop and apply a kind of integrative freshwater complexity indicator (i.e. quantifying landscape heterogeneity based on composition, configuration, and connectivity of freshwater ecosystem types). Indicators exist for individual ecosystem types (e.g. shoreline index, floodplain complexity index) but not yet at the catchment/landscape scale (a meta-ecosystem indicator).
- Standardization of all attributes (indicators) from 0-1.

Screening of pressures, stressors and risks as well as selection of potential indicators and metrics for different water body types (Step 4a)



Background Information

Level 1 assessment background information is collected for the study area on factors, such as:

- Population density,
- Human wealth (e.g. population density),
- GDP,
- Catchment topography, elevation,
- Land use/land cover types,
- Protected areas,
- Biogeographic region,
- Lentic surface waters (i.e., standing water bodies such as lakes, reservoirs): area, density, volume, perimeter-to-area ratio, and diversity (typology),
- Lotic surface waters (i.e. running waters, such as streams and rivers): network density, stream order, and diversity of types,
- Inland palustrine wetlands: area, diversity (typology), and conversion rate,
- Relative composition, spatial configuration, and degree of connectivity of water bodies (aggregated landscape complexity),
- Nitrogen- and phosphorous-deposition.

General Screening (Level 1) and in-depth screening (Level 2)

Depending on the needs and time frame, a two-tiered desktop screening process is proposed. Level 1 provides a desktop level assessment based on a limited number of stressors. Level 2 is a more detailed, but still desk based assessment. When shifting from Level 1 to Level 2, the diagnostic capability increases, but also the costs and technical requirements. Overall it will increase the confidence towards the assessment of the water quality status.

Assessment of pressures and stressors

Table 4.2 demonstrates the suggested screening steps of Level 1 and 2, i.e. screening of lakes and reservoirs, wetlands, wadeable and non-wadeable rivers on the example of selected stressors and their potential indicators.

Table 4.2 Checklist of screening Level 1 and 2 monitoring and assessment

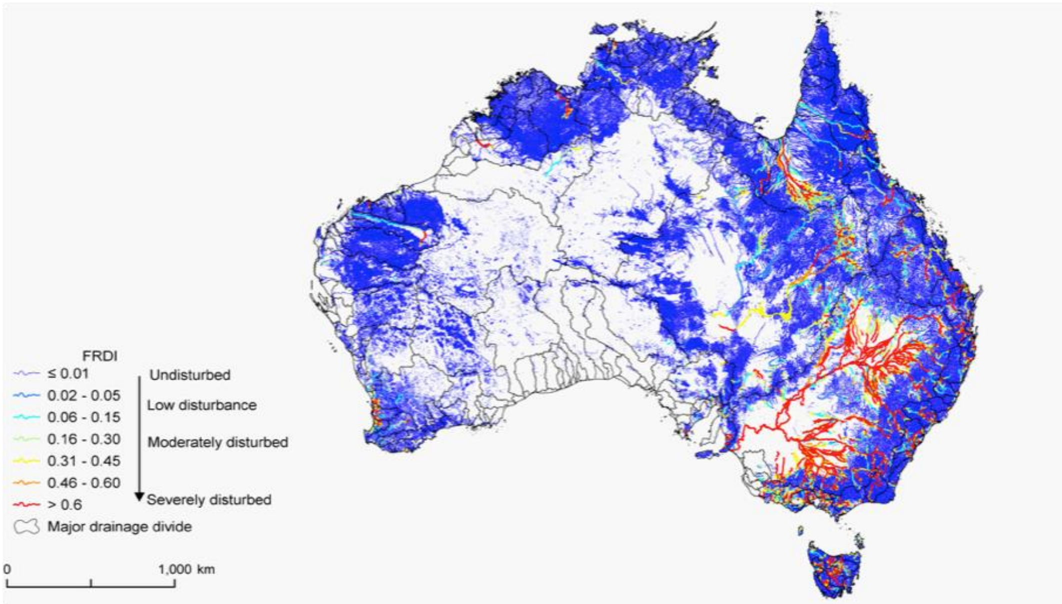
CV is the coefficient of variation of the annual inflows (Coefficient of Variation (CV) = Standard Deviation (SD)/mean). Suggested spatial resolution of the maps and/ or GIS layers used is 0.5 degrees (latitude/longitude).

Background info		**Lakes and reservoirs**	**Wetlands**	**Rivers (wadeable)**	**Rivers (non-wadeable)**
Screening Level 1					
Assessment of pressures and stressors	Catchment landuse	Especially % urban, % industrial area, % intensive agriculture, % mining, % forest cover (0.5 degree); domestic and industrial discharges			
	Water Uses	Intensity of water uses: shipping, recreation, fishing and water withdrawal			
	Catchment population density	Mapped at subcatchment scale at least (e.g. 0.5 degree) (note – could use light intensity)			
	Dams - barriers	Total number of dams and storage capacity in catchment			
Screening Level 2					
Assessment of pressures and stressors	Riparian/fringing landuse	Especially % urban, % industrial area, % intensive agriculture, % mining, % forest or natural vegetation cover – eg 500m buffer plus buffer on catchment streams (remote sensing, aerial photography, field assessment); domestic and industrial		Especially % urban, % industrial area, % intensive agriculture, % mining. % forest or natural vegetation cover – e.g. 500 m riparian buffer (remote sensing, aerial photography, field assessment); domestic and industrial discharges	
	Water Uses	Intensity of water uses: shipping, recreation, fishing and water withdrawal			
	Riparian/fringing population density	Mapped at fine scale – population living within 500 m buffer plus buffer on catchment		Mapped at fine scale – e.g. population living within 500 m buffer zone	
	Dams - barriers	Measures of flow alteration (e.g.% extraction, changes in ecologically relevant flow			
	Fragmentation index	Indices of fragmentation – e.g. 'swimmable distance', 'wetland discontinuity' (Vörösmarty et al., 2010), barriers upstream and downstream			
Other important environmental factors	Rainfall	Annual rainfall in catchment, rainfall CV			
	Hydrology	Annual discharge, annual CV of inflows, depth range, intermittency		Annual discharge, annual CV, seasonality, predictability, dry spell duration	
	Topography	Elevation, slope			
	Geology/soils	Major soil/rock types (to detemine natural influences on WQ – e.g. conductivity, hardness, Total Suspended Solids (TSS)			
	Temperature	Max/min monthly air temperature; temperature range			

A key goal is not only to assess the present environmental state at the catchment level but also to assess the direction and rate of change, especially on those pressures and stressors that represent a risk for deterioration of ecosystem health. Indicators may include human population growth, economic development, land-use change, climate change related alteration in precipitation, temperature and flow pattern, expected changes in water use, large-scale water and other infrastructure development, among others.
Finally, a key challenge is to define scale and distribution of the pressure information. Because the catchment is considered as the basic unit for assessment and subsequent management, information must be as spatially-explicit as possible, to be of practical management value.

»Figure 4.5 provides an example for the assessment of pressures acting on rivers in Australia.

Figure 4.5: An example of a map of stressors to freshwater ecosystems based on flow alteration disturbance from the presence of dams, diversions and levees. Source: Stein et al. (2002; updated 2010).



Causal assessment: screening for critical stressors
The specific causes of undesirable biological effects in ecosystems often are not known. However, the cause(s) of an effect must be determined before it can be remediated by appropriate management actions (Norton et al. 2015; Suter et al. 2010). Determining the likely cause of a certain biological condition relies on weighting the body of evidence for each cause that is considered (Suter and Cormier 2011). For each of these candidate causes, evidence is sought that documents six expected characteristics of causation:

1. co-occurrence of the cause and the effect,
2. sufficient exposure to cause the effect,
3. antecedent cause-effect events,
4. specificity of symptomatic alterations in the biological assemblage or organisms,
5. precedence of the cause effect in time relative to the effect, and
6. interaction of the cause with the ecological or biochemical processes that produce the overt effect (Cormier et al. 2010).

To facilitate causal assessments, the United States Environmental Protection Agency (US EPA) developed the Causal Analysis/Diagnosis Decision Information System (CADDIS). CADDIS is an online support system to help users conduct causal assessments, primarily in stream ecosystems. It provides a logical, step-by-step framework for Stressor Identification (SI) based on the US EPA´s SI Guidance Document (Cormier et al. 2003), as well as information and tools that can be used in these assessments and example cases.

The causal assessment process has three general stages

Stage 1: Formulate the problem by defining the effect and developing a list of candidate causes for evaluation. This is accomplished by carefully describing the effect that has initiated the causal assessment (e.g., unexplained absence of an important fish species) and its temporal and spatial extent, and gathering available information on the situation and potential causes. The involvement of decision-makers and stakeholders is particularly important when defining the scope of the investigation and ensuring that the list of candidate causes is complete. The outputs of this initial step are

i. explicitly defined effects,
ii. a list of candidate causes,

iii. a conceptual model that shows possible cause and
iv. defined effect relationships.

Stage 2: Generate evidence for and against each proposed candidate cause using all available and relevant data. Evidence may come from observations, experiments, or general knowledge of processes or mechanisms. Virtually everything that is known about an impaired ecosystem is potentially useful at this stage. For example, useful data may come from chemical analysis of effluent, water, sediment, and tissue samples; surveys of the presence, abundance, and conditions of organisms; toxicity tests; necropsies; habitat measurements; climatic and hydrologic records; and biochemical measurements. These data do not in themselves constitute evidence of causation. Evidence of causation shows an association between what is expected to occur given that the hypothesized candidate(s) caused the effect. For example, a chemical concentration in the water at the site at the time of a catfish kill is expected to be greater than the concentration that causes standard test fish to die in laboratory test. This is an association between site concentration and concentrations sufficient to cause a similar effect in a laboratory setting. It is not proof of causation in the environment because the field and test species have different susceptibilities, the water concentration of the sample may be greater than when the fish died, the chemical may not be bioavailable in the field, etc. For these reasons, many types of evidence and comparison among different candidate causes are needed to form consolidated conclusions.

Stage 3: Form conclusions that best explain what caused the effect. Effective conclusions provide clear reasoning that informs decision making for resolving an environmental problem. The investigator weights the body of evidence for each candidate cause and compares among causes to which cause or causes are best supported by the evidence. In straightforward cases, the process may be completed in linear fashion, discarding impossible and unlikely causes until one or a few causes remain. In more complex cases, interactions among causal pathways will provide a better explanation. For example, low Dissolved Oxygen (DO) may be the cause of a fish kill but only during warm weather. When the assessment is inconclusive, the results can be used to identify additional data needs and to focus and refine future assessments. Once the cause or causes are identified, managers can pursue appropriate management actions to control, restore, or protect biological condition. Ideally, the effectiveness of those management actions will be monitored.

The following list may be used in identifying and characterizing the most common and strongest stressors and also states examples of potential indicators (» Table 4.3).

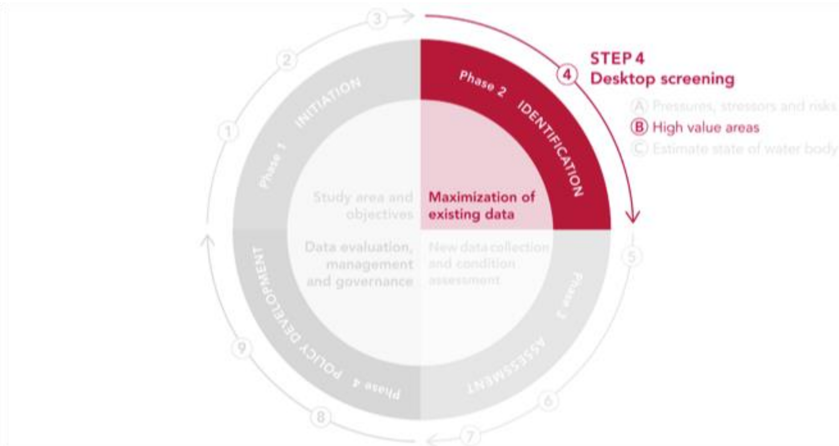
Table 4.3: General description of stressors, their manifestation and examples of potential indicators

Stressors	Manifestation	Examples of potential indicators
Water infrastructure	Horizontal and vertical discontinuities: dams, dykes, sluices, barrages, groins, weirs, inlet and outlet structures, canals, turbines, interbasin transfers	Number and size of structures, their discharge capacity, length of dykes, reduction of floodplain area, area of groin fields, length of canals and tunnels
Flow alteration	Water withdrawals, discharges, hydro power operations, interbasin transfers, reservoir release, irrigation and flood control	Rate and duration of withdrawals and discharges, number and volume of reservoirs within the basin, hydropower capacity and production in kWh/a, change of shape and volume of the natural hydrograph
Modification of aquatic habitat	Sedimentation, river training, dredging, mining, eutrophication, navigation, unconventional fishing gears	Depth of sediment deposition, turbidity, volume of excavated materials (sand, gravel), length of modified water course, nutrient loads and concentrations, frequency of ship passages and tonnage

Overexploitation	Overfishing, hunting, aquaculture, excessive water withdrawal, gravel and sand mining	Yields as t/year for different species, number of mining sites and volume of excavated material overexploited sites/area as percentage of water bodies
Biological water pollution	Occurrence of alien species via migration or human interference, aquaculture escapes	Number of invasive species, alteration of distribution of species, decline of native species, percentage of water bodies affected
Chemical water pollution	Agricultural (e.g., pesticides, nutrients), industrial (e.g., heavy metals, persistent organic pollutants), emerging pollutants (e.g., endocrine disruptors)	Shares of treated/untreated sewage, concentrations and loads of chemical elements and compounds in discharged (waste) water and in the recipient water bodies, waste water as percentage of natural flow, share of airborne deposition, Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD) of the (un) treated waste water
Thermal water pollution	Cooling water discharge and release of cool water from reservoirs	Ambient water temperature and temperature of cooling water, discharge of cooling water and released cool water related to the temperature of natural flow

If possible, in this step some attention should be paid to future development stressors, especially those which may cause the risk for deterioration of the ecosystem health. Socio-economic developments in terms of expected increase of population, economic growth, expected changes in land and water use may be considered. Effects on ecosystem health might be expected from (strong) growth of population, (strong) growth of economic activities like industrialization, shipping, fishery, recreation or large scale infrastructural plans, such as land reclamation or hydropower developments.

Identification of high-value areas and water bodies or ecosystems for protection (Step 4b)



The EU WFD defines 'high status' as the biological, chemical and hydromorphological conditions associated with no or negligible to very low human pressure. This is often also called the 'RC' as it is the best status achievable. It can be defined either spatially (reference locations) or temporarily (reference period for a specific water body) or both. These high-status conditions are ecosystem type-specific and differ among biogeographic regions.

The assessment of the state of an ecosystem is based on the extent of deviation from these RCs (or high status conditions): 'Good status' means 'slight' deviation, 'moderate status' means 'moderate' deviation, and so on. The definition of ecological status takes into account specific aspects of the Biological Quality Elements (BQE), for example "composition and abundance of aquatic flora" or "composition, abundance and age structure of fish fauna" (see EU WFD Annex V Section 1.1 for a complete list and definitions).

Desktop Study "High Status"

Stressors: For each stressor category the upper threshold level that separates the high status of ecosystems will be defined based on desktop studies and on site-specific data sources, if available. Values will be standardized (0-1) to allow comparison among stressors as well as across systems and regions. Finally, a multi-stressor index that integrates the different stressors (weighted or unweighted) can be calculated for each catchment unit.

Catchment land-cover and land-use: The relative proportion of urban area and cropland (intensive land use categories) may be used to identify a high status of the ecosystem health. The threshold values must be calibrated for individual catchments and ecosystem types. Overall, however, high status areas will be areas where urbanization and irrigated croplands are almost completely absent; although the threshold value may depend on agricultural intensity and urban development.

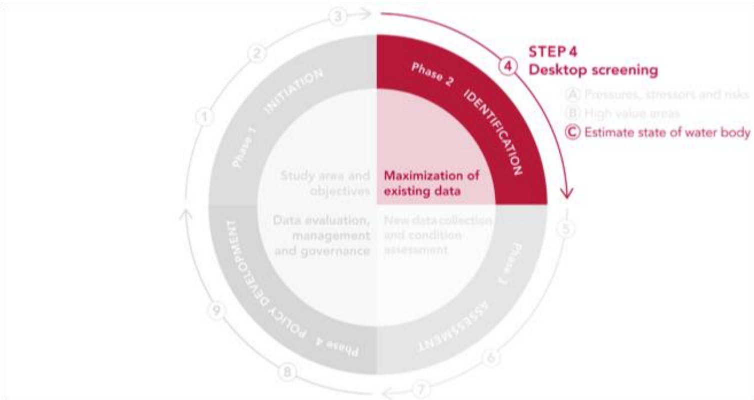
Flow regime and water quality: High status systems include free-flowing river networks (for some accepted definitions see: Nilsson *et al.* 2005; Zarfl *et al.* 2015), lakes with near-natural water level regimes, and wetlands with a near-natural groundwater-surface water regime. The oxygen, thermal, nutrient, pH and sediment regimes will depend on the degree of longitudinal or/and lateral fragmentation of the water course and the geomorphic regulation; therefore riparian land use will also need to be assessed. High status systems are characterized by fringing riparian and floodplain systems that exhibit a high proportion of near-natural land cover and an unimpeded lateral degree of connectivity.

Hydromorphological conditions: For lakes and rivers, a simple index of complexity may be used to assess the near-natural status as well as the deviation from it (O´Neill and Thorp 2011). It may consider shoreline length (to area ratio), braiding or sinuosity indices, and the proportion of near-natural shore areas. For wetlands, the hydrogeomorphic index (Brinson 1983) still serves as a valuable approach to assess their status.

Ecosystem diversity and/or complexity: A landscape-based index that considers the composition, configuration and connectivity of freshwater ecosystem types can be calculated for each catchment unit (i.e. different spatial scales). This index can be compared to the historic ecosystem diversity based on reference/historic material such as maps. However, ecosystem diversity remains one of the most neglected components of biodiversity. Development and application of such an index, or the adaptation of landscape diversity indices (e.g. Soininen *et al.* 2015) for the required purpose would as a first step, need to include and calculate the number of ecosystem types, the relative proportion and the eveness of ecosystem types, and the functional properties of ecosystem types; in line to the assessment of different components of species diversity.

Biological conditions: High status areas can be defined using the KBA approach developed by International Union for Conservation of Nature (IUCN) within the frame of BioFresh (http://www.freshwaterbiodiversity.eu). KBAs are places of international importance for the conservation of biodiversity through protected areas and other governance mechanisms. This may include Ramsar listed wetlands, rivers or other wetlands in National or State parks, or in other kinds of protected areas. As the building blocks for designing the ecosystem approach and maintaining effective ecological networks, KBAs are starting points for a systematic conservation planning at the landscape/catchment level (e.g. Le Saout *et al.,* 2013; Barrios *et al.,* 2014). Governments, intergovernmental organizations, Non-Governmental Organizations (NGOs), the private sector, and other stakeholders can use KBAs as a tool for identifying and delineating networks of internationally important sites for conservation. Information on KBAs is already available for several continents and will most likely be available within the coming years - at high spatial resolution - at the global scale. An example of a KBA for freshwater ecosystems is presented in »**Error! Reference source not found.**. High value areas can also be identified nationally using simple, standard criteria (e.g. number of or proportion of threatened and endemic species, or areas of high biodiversity), based on their importance in maintaining species populations (Holland *et al.*, 2012).

Estimation of the present state of each water body (Step 4c)



This step consists of an inventory of available data concerning the present state, which might be accessible in reports of water and environmental authorities and other government technical departments, or specific research projects by (international) research institutes, universities and NGOs (see »Table 4.4 for relevant biological, physico-chemical and hydromorphological indicators). Sometimes, stakeholders e.g. fishermen may provide relevant information concerning the (change in) quality status. This inventory will also provide an overview of indicators for which data are scattered or entirely lacking.

Based on the information about the stressors and the inventory of data about the present states, a first qualitative estimate can be made of the water quality categories of the systems considered. In this identification phase, the attribution of a water quality category to a system can be based on the qualitative description of the freshwater ecosystem health unless quantitative data was available.

Based on the overview of quality states and future developments and threats, specific goals have to be set for improving or preventing the deterioration of the ecosystems considered. A decision has to be made on which freshwater ecosystems need the highest priority for a more detailed assessment.

Table 4.4: Possible biological, physico-chemical and hydromorphological indicators.

System feature	Indicator categories	Indicators
Biological	Fish	Sensitive taxa; Relative richness; Size/age structure; Disease incidence; Alien species; Trophic structure; Life history traits; Reproductive traits
	Invertebrates	Sensitive taxa; Relative richness; Size/age structure; Life history traits; Sensitive taxa; Trophic structure; Community composition
	Algae	Sensitive taxa; Community composition; Algal biomass
	Macrophytes	Sensitive taxa; Taxa composition; Abundance
Physico-chemical	Metabolic rate	Oxygen (DO, BOD, COD); Temperature; pH; Light penetration (Secchi depth); Conductivity/salinity; Microbial pollutants (*E. coli*, total coliform count)
	Trophic state	Nutrients (N, P, NH_4, NO_X, soluble P); Chlorophyll *a* (Chl-*a*)
	Toxicants	Heavy metals (Cd, Hg, Cr, Cu, etc.); Pesticides; Other organic pollutants (oil, phenol, PCBs, endocrine disruptors)
Hydromorphological	Aquatic habitats	Colonisable substrates; Substrate condition; Velocity and depth variability; Sediment deposition; Channel flow; Habitat diversity;

		Aquatic vegetation; Off channel aquatic habitats
	Riparian habitats	Bank stability; Bank vegetative cover and protection; Human influence

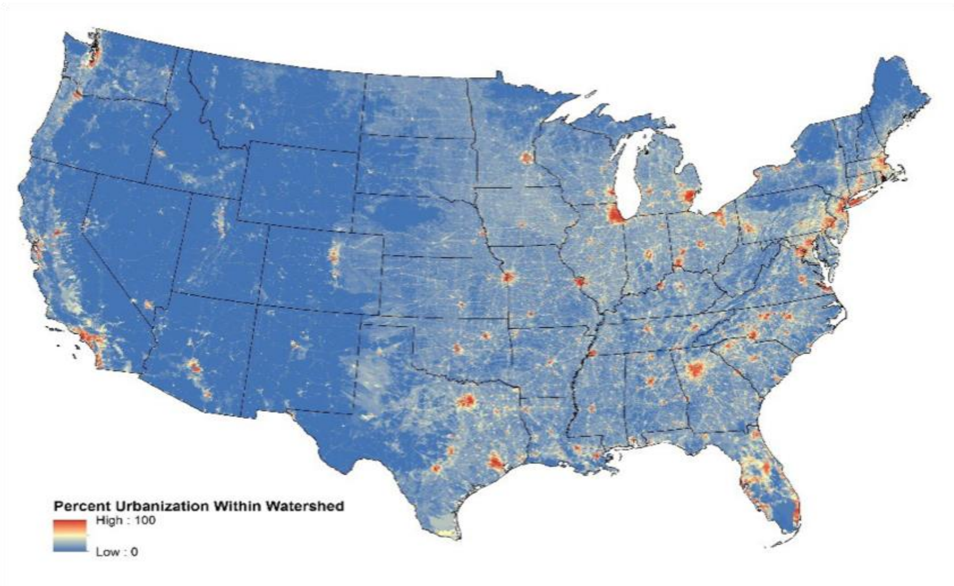
Using an index of watershed integrity to screen for pressures, stressors and state of inland water bodies

Watershed 'integrity' may be defined as the capacity of a watershed to support and maintain the full range of ecological processes and functions essential to the sustainability of biodiversity and of the watershed resources and services provided to society (Flotemersch et al., 2015). In recent application as an assessment tool, risk factors (i.e. human-related alterations or stressors) that have been explicitly shown to interfere with and degrade key functions in water bodies are identified and then used to model and map watershed integrity. An advantage of the IWI approach is that the index can be readily deconstructed to identify the factors influencing index scores, thereby directly supporting the strategic adaptive management of individual components that contribute to watershed integrity. Moreover, the approach can be iteratively applied and improved as new data and information become available. Once constructed, the maps can be used to inform on the pressures, stressors and state of inland water bodies.

As an example of this approach, researchers are calculating an Index of Watershed Integrity (IWI) for all watersheds of the continental United States. To facilitate this, six key watershed functions were identified (viz. hydrologic regulation, regulation of water chemistry, sediment regulation, hydrologic connectivity, temperature regulation, habitat provision) along with the specific risk factors, or stressors, which impact them. Values for these risk factors are being used to evaluate watershed integrity by combining the integrity of the six watershed functions, where the integrity of each is based on the relative presence of specific stressors. Coverages for calculating the IWI in the U.S. are derived from StreamCat, which is a centralized dataset of watershed characteristics (Hill et al., 2016 and https:// www.epa.gov/national-aquatic-resource-surveys/streamcat). »

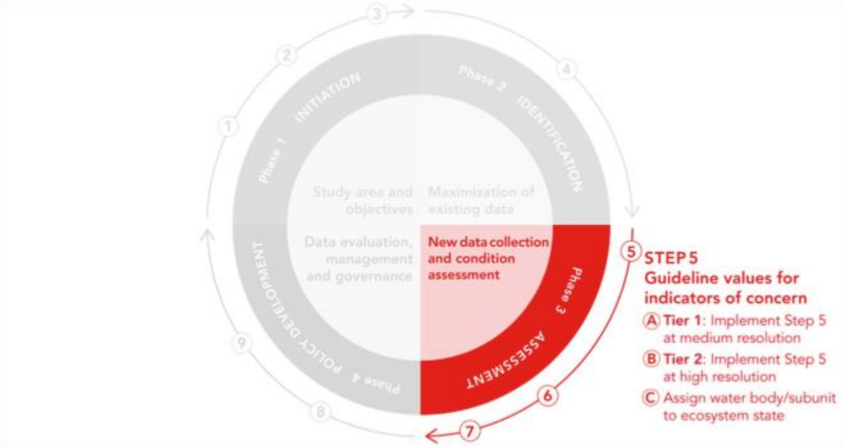
Figure 4.6 shows percent urbanization from StreamCat for the entire US. Coverages such as these can be used to inform on pressures, stressors and state of inland water bodies.

Figure 4.6: Example of a US map created using StreamCat data on percent urbanization within a watershed.
Source: Hill et al. (2016).



Guideline values (narrative and/or numerical) for indicators of concern (Step 5)

N.B. This step matches the ‘Set ecological status thresholds and targets’ step in the revised Framework (See Preface).



General description

The ecological status, the quality of the structure and functioning of freshwater ecosystems are characterized by biological, physico-chemical and hydromorphological factors. Hence, a guideline for the quality of freshwater ecosystem should specify all three types of indicators preferably with their respective numerical values. In »Chapter 3 existing narrative and numerical criteria for fresh water ecosystems are discussed. As concluded in »Section 3.2.2 the guidelines in Australia/New Zealand, the European Union (EU) and the United States of America (US) provide most up-to-date and innovative approaches for integrated ecosystem assessment including biological, physico-chemical and hydromorphological indicators.

The EU WFD may be considered as a framework, which has most detailed and itemized ecological objectives for different type of water bodies. In »Annex 4 of this report the normative definitions of ecological status classification as specified in the EU WFD are presented as well as the biological, hydromorphological and physico-chemical quality elements. In the last decade in the US new methods have been introduced for biological assessment to support water quality management (»Annex 3).

The Australian/New Zealand guidelines also provide a number of biological assessment methods and physico-chemical criteria as described in »Chapter 3. Moreover in Australia an aquatic ecosystems toolkit for identifying High Ecological Value Aquatic Ecosystems (HEVAE) was developed. In some other countries initiatives have been taken to develop guidelines for aquatic ecosystem and to incorporate biological indicators in monitoring programmes, e.g. South-Africa (DWAF, 1996a), Indonesia (MoE, 2008) and Brazil (ANA, 2012).

Establishing guideline values requires a clear definition of the level of protection that is aimed at. In this volume, four principal water quality categories are suggested for subdividing the quality continuum. These categories are associated with different states of ecosystem integrity and health of the respective freshwater system. See »Section 4.1.4 for the definition of these categories.

It is important to emphasise that, although these are described as WQGs, measurements of the physical and chemical properties of water alone cannot provide a guarantee that biodiversity is protected or key ecosystem processes are intact. Water quality measurements may reflect a high standard for aquatic life, however, ecological integrity can still be impacted because of changes to the physical habitat (including flow regime) or because of some unmeasured toxicant, or due to an episodic event with poor water quality prior to sampling. In order to overcome these problems, it is imperative to focus on biological parameters as well as hydromorphological and physical-chemical indicators simultaneously.

A key element of an assessment is on biological parameters such as flagship and indicator species, based on existing knowledge of their distribution and their specific traits. Using species traits allows a function-based assessment of freshwater ecosystems. Concurrently, the composition of biological assemblages reflects very well the environmental state of an ecosystem, and serves as an integrator across multiple stressors and various time periods. Therefore, biological assemblages are robust and at the same time sensitive indicators for assessing ecosystem health. »Figure 2.13 highlights the schematic relationship between the proposed classification of water quality status and the ecological condition gradient. This methodology is described in more detail in Annex 3.

The multi-level approach presented in the Framework distinguishes between essential and advanced options (Tier 1 and 2) as far as number and type of indicators and monitoring efforts are concerned. This procedure allows also staged development and phased introduction of standards and their respective enforcement mechanisms similiar to the “ladder approach” advocated in connection to SDG achievement monitoring (UN-Water, 2015). Selecting what and how frequently to monitor is a decision of the implementing agency. Achieving and maintaining high ecological status for freshwater bodies implies an elaborate and regular monitoring of selected indicators and additionally, the subsequent implementation of remedial actions should they be needed, as well as their success evaluated.

Some indicators proposed below are valid across all freshwater ecosystems; others will need to be distinguished as specific threshold values considering the context (climatic and/or geomorphic zones). Not all of the indicators will need to be measured regularly. All in all, basic and advanced field and laboratory water physico-chemical indicators will be considered, as well as hydromorphological and biological indicators, including toxic constituents, system variables, process-based indicators, key stressors, and finally novel indicators based on results and future recommendations of ongoing research.

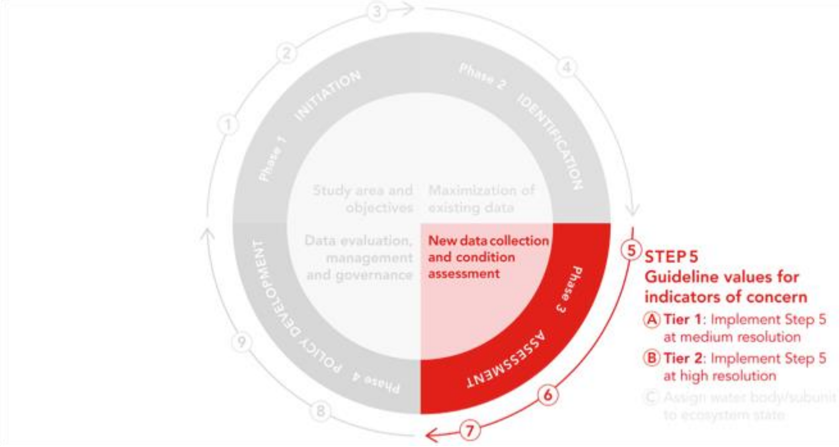
While Steps 1 to 4 are desktop assessments, in Step 5 the field sampling is prepared which would be required through a tiered approach: for Tier 1 (Step 5a) a limited field assessment for a relatively small number of selected indicators is proposed, while in Tier 2 (Step 5b) a detailed field assessment with a more extensive number of indicators should be undertaken. From Tier 1 to Tier 2 the diagnostic capability, but also the costs, technical requirements and confidence towards the assessment of status increase.

The Framework supports the assessment of the present state of ecosystem health/integrity; however, it also calls for actions. For example, Category 1 freshwater ecosystems require protection, while Category 4 ecosystems ask for fundamental restoration efforts and possibly engineering solutions. Categories 2 and 3 imply the potential need for improvement (depending on the societal/political aspirations agreed on, as outlined further up in this section), in particular if the ecosystem health status is close to the “dashed line” benchmark (as shown in »Figure 4.1 and »Figure 2.13).

In the following sections 4.5.2 - 4.5.4, possible indicators are listed for the biological, physico-chemical, and hydromorphological assessments for both, Tier 1 (Step 5a) and Tier 2 (Step 5b) monitoring levels.

Biological guideline values (Steps 5a and 5b)

Threshold values for biological indicators are more difficult to establish than physical/chemical ones, because the values of biological indicators strongly depend on the type and location of the ecosystem considered. Even within certain classes of freshwater ecosystems (lakes, rivers, wetlands), there are many natural differences in physical, chemical and morphological



conditions and the natural situation of an ecosystem. Moreover, the natural conditions may change over time by natural processes such as erosion or ecosystem succession.

A serious attempt to classify the quality of biological indicators for rivers, lakes, transitional waters, coastal waters and artificial waters is undertaken in the EU WFD (See »Annex 4). Based on the general description in the EU WFD member countries should establish more specific classification criteria for the relevant indicators. As an example in »Table 4.5 a description of three levels of a status for fish in lakes is presented as established in the United Kingdom (UK). The same types of descriptions are available for phytoplankton, macrophytes and invertebrates. However, using this approach requires information about the natural/undisturbed system. These RCs may be estimated by a description of the ecological situation in undisturbed systems in the same climate zone and comparable hydrogeomorphic and chemical conditions.

Table 4.5: Lake – Description of status for fish.
Source: UK TAG (2013[51]).

High status	Good status	Moderate status
Species composition and abundance corresponds totally or nearly totally to undisturbed conditions. All the type-specific sensitive species are present. The age structure of the fish communities show little sign of anthropogenic disturbance and are not indicative of a failure in the reproduction or development of a particular species.	There are slight changes in species composition and abundance from the type-specific communities attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements. The age structure of the fish communities show signs of disturbance, attributable to anthropogenic impact on physico-chemical or hydromorphological quality elements and, in a few instances, are indicatives of a failure in the reproduction or development of a particular species, to some extent that some age classes may be missing.	The composition and abundance of fish species differ moderately from the type-specific communities attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements. The age structure of the fish communities show major signs of disturbance, attributable to anthropogenic impact on physico-chemical or hydromorphological quality elements, to the extent that a moderate proportion of the type-specific species are absent or of very low abundance.

Table 4.6 and »Table 4.7 list the proposed indicators and monitoring recommendations at Tier 1 and Tier 2 levels for biological assessment of freshwater bodies. The colour coding refers to primary (green) intermediate (yellow) and advanced (red) level indicators. Moving from “green” towards “red” implies increasing costs and technical requirements. »Box 4.3 states further

[51] UK Technical Advisory Group on the Water Framework Directive (2013). Final Recommendations on New and Updated Standards. www.wfduk.org/sites/default/files/Media/UKTAG%20Final%20recommendations%20on%20biological%20stds_20131030.pdf

examples.

According to the Water Framework Directive the methods for assessment of the ecological status are based on the national systems. However for harmonization of final class boundaries for biological communities the process of intercalibration has been started among European Member States based on Guidance Document No. 14[1] and No. 30[2]. The results on intercalibration have been published in 2008 and 2013 as a European Commission (EC) Decision.
Technical Reports on Intercalibration are available for coastal and transitional waters[3] as well as for rivers[4] and lakes[5].

1https://circabc.europa.eu/sd/a/61fbcb5b-eb52-44fd-810a-63735d5e4775/IC_GUIDANCE_FINAL_16Dec2010.pdf
[2]https://circabc.europa.eu/sd/a/5aee6446-276c-4440-a7de-0d4dec41ed4b/IC_manual_2015_to%20be%20published.pdf
[3]http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/10473/1/3010_08-volumecoast.pdf
4http://www.apambiente.pt/dqa/assets/intercalibration-2003-2007-technical-report---rivers.pdf
5http://www.apambiente.pt/dqa/assets/intercalibration-2003-2007-technical-report---lakes.pdf

Box 3 ‘No net loss’ policies for wetland extent.

Table 4.6: Proposed biological indicators for Tier 1 level monitoring. Colour coding: primary (green), intermediate (yellow) and advanced (red) level indicators.

TIER 1 MONITORING					
		Lakes and reservoirs	**Wetlands**	**Rivers (wadeable)**	**Rivers (non-wadeable)**
Indicator group	**Indicator (one or more)**	**Method**			
Fish	Assess local fish catches (e.g. fishers; local markets) – note absence of fish = 'dead zone'				
	Sensitive taxa	Ratio of sensitive/tolerant families or genera relative to total (e.g. from Fishbase)			
	Relative richness	Diversity of taxa, where possible relative to regional pool (derived from Fishbase and/or local knowledge)			
	Size/age structure	Size (or biomass) frequency distributions (evidence of mortality of older individuals, and/or only presence of juveniles – potentially from elsewhere – and/or overfishing)			
	Disease incidence	% fish with lesions, tumours (evidence of stress and exposure to toxicants)			
	Alien species	% introduced or non-native fish (these often do better in systems that are disturbed by human activity – especially flow alteration, eutrophication)			
	Trophic structure	Functional feeding groups			
	Life history traits	N/A for Tier 1			
	Reproductive traits	Metric of recruitment where feasible – eg. fish and clams			
Invertebrates	Assess local catches of prawns/shrimp /crabs/snails/mussels. Rapid assessment techniques for non-commercial species				
	Relative richness	Diversity of taxa, where possible relative to regional pool (derived from local knowledge)			
	Size/age structure	Size (or biomass) frequency distributions (evidence of mortality of older individuals, and/or only presence of juveniles – potentially from elsewhere – and/or overfishing)			
	Life history traits	N/A for Tier 1			
	Sensitive taxa	Presence or absence if known			
	Trophic structure	Functional feeding groups			
	Community composition	N/A for Tier 1			
	Invasive species prevalence	Number and prevalence index of invasive species			
Algae	Note – some indication of algal biomass may come from measures of water clarity				
	Taxa composition	N/A for Tier 1			
	Sensitive taxa	N/A for Tier 1			
	Algal biomass/cover	Mean Chlorophyll *a*; presence of blooms (only if no normal manifestation of seasonal succession)		Mean Chlorophyll *a*; excessive algal cover	Mean Chlorophyll *a*; exessive algal biomass
Macrophytes	Invasive surface macrophyte presence	Salvinia, hyacinth, etc.			Salvinia, hyacinth, etc.
	Invasive emergent macrophyte presence	Invasive species of reed	Cattails, invasive species of reed		

Color coding	
green	primary (relatively easy and cheap)
yellow	intermediate
red	advanced (more complex, cost and

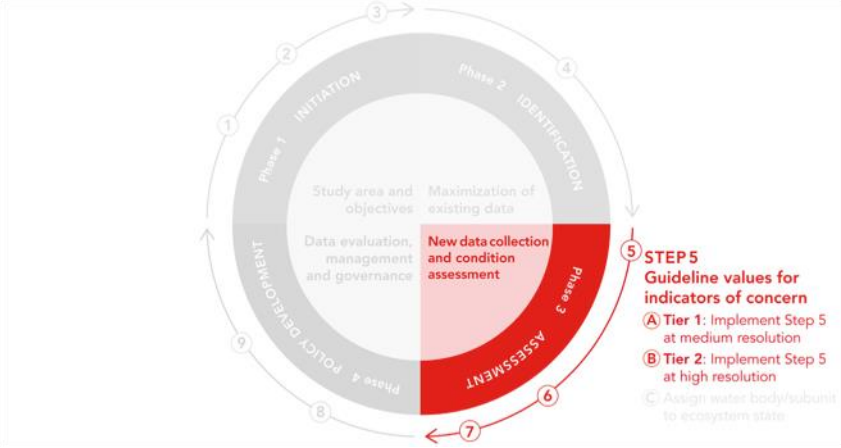
Table 4.7: Proposed biological indicators for Tier 2 level monitoring.

Colour coding: primary (green), intermediate (yellow) and advanced (red) level indicators. O/E represents the ratio of the number of families of invertebrates observed at a site to the number of families expected; Fishbase: http://www.fishbase.org; IBI = Index of Biotic Integrity (Karr, 1981); OMNIDIA = software for taxonomy, calculation of diatom indices and inventories management (Lecointe et al., 1993).

TIER 2 MONITORING					
		Lakes and reservoirs	**Wetlands**	**Rivers (wadeable)**	**Rivers (non-wadeable)**
Indicator group	**Indicator (one or more)**	**Method**			
Fish		Pelagic trawls, gill nets or littoral sampling by seine net or electofishing by boat	Seine or fyke netting or back-pack electrofishing; traps	Seine netting or back- pack electrofishing	Set gill, fyke or seine nets, or electrofishing by boat
	Sensitive taxa	Could be qualitative sampling but needs to be standardized for specific habitat(s) and method(s) – Ratio of sensitive/ tolerant familes or genera relative to total (e.g. from Fishbase)			
	Relative richness	Standardized composite netting and/or electrofishing, by habitat(s) and method(s) – development of predictive models based on environmental factors to estimate expected values per unit effort (O/E)			
	Size/age structure	Standardized composite netting and/or electrofishing, by habitat(s) and method(s) – size (or biomass) frequency distributions			
	Disease incidence	Could be qualitative sampling but needs to be standardized for habitat(s) and method(s) – % fish with lesions, tumours			
	Alien species	Could be qualitative sampling but needs to be standardized for habitat(s) and method(s) – % introduced fish			
	Trophic structure	Standardized composite netting and/or electrofishing, by habitat(s) and method(s) – % predators; % benthic and/or pelagic feeders			
	Life history traits	Standardized composite netting and/or electrofishing, by habitat(s) and method(s) – % migratory species			
	Reproductive traits	Standardized composite netting and/or electrofishing, by habitat(s) and method(s) – % nest builders, % vegetation spawners, % broadcast spawners			
Invertebrates		Zooplankton trawls; benthic grabs; littoral sweep net; shrimp traps	Littoral sweep net samples; shrimp traps	Kick net (benthic sampler)	Zooplankton trawls; benthic grabs; littoral sweep net
	Relative richness	Development of predictive models based on reference from similar environmental settings to estimate expected values per unit effort (O/E)			
	Size/age structure	Size (or biomass) frequency distributions of shrimps, crabs, mussels (evidence of mortality of older individuals, and/or only presence of juveniles – potentially from elsewhere – and/or overfishing)			
	Life history traits	Age structure of selected taxa – can be very useful for species with high longevity (e.g clams and mussels)			
	Sensitive taxa	cladoceran to copepod ratio	e.g. %EPT (mayfly, stonefly, caddisfly taxa); IBI		
	Trophic structure	Zooplankton metrics, size and fecundity	% grazers, % predators, % fine-particle feeders		
	Community composition	can also have a reference type approach applicable to some stuations more than others	Development of predictive models based on reference from similar environmental settings to estimate expected composition (O/E)		
	Invasive species prevalence	Number and prevalence index of invasive species			
Algae		Phytoplankton tows	Epiphytes from vascular plants or rocks; microscope slides	Algal scrapes from rocks or extraction from sand	Phytoplankton tows; algal scrapes from logs or littoral rocks; microscope slides
	Taxa composition	% cyanobacteria; % toxic cyanobacteria: % diatoms		% diatoms,	% cyanobacteria; % toxic cyanobacteria: % diatoms
	Sensitive taxa	?	?	Diatom indices (e.g. OMNIDIA)	
	Algal biomass/cover	Chl-*a* / m³; % cover of blanketing filamentous	Chl-*a* / m² substrate; monthly	Chl-*a* / m²	Chl-*a* / m² substrate or

		algae (shallow lakes); monthly fluctuations of Chlorophyll *a* in the growth season	fluctuations of Chlorophyll *a* in the growth season	substrate; % cover of blanketing filamentous algae	m^3 water
Macrophytes	Invasive emergent macrophyte presence	cover of invasive species (areal or percent Vegetation infested). Lakes and wetlands espepcial, but maybe also for rivers			

Physico-chemical guideline values (Steps 5a and 5b)



Concerning physical/chemical objectives, a large number of quantitative water quality indicators/standards have been established in a number of countries and regions. Quality standards for oxygen levels, nutrients and pH have been derived from natural levels and studies about the effects on the ecosystem of decreased oxygen concentration, the impact of increased nutrient concentration and effects of pH changes. Water quality standards for pollutants are based on ecotoxicological experiments in laboratories mostly, and include both acute and chronic toxicity tests. The most used taxa in these laboratory tests are algae, zooplankton (daphnia) and fish. Safety factors are used to establish water quality standards. The magnitude of the safety factor may depend on the number of available toxicity data and the defined protection level. If data are available about persistence and bioaccumulation these can also be taken into account, as well as data from field studies. »Table 4.8 provides freshwater benchmark values for physical and chemical indictors which are indicative of high ecosystem integrity and extreme impairment, respectively. The first benchmark value will separate ecosystems of high integrity (Category 1) from other ones. The second benchmark demarcates the lower end of the quality continuum where water quality severely constrains the existence of most forms of multicellular life and ecological structure and function (Category 4). Values to demarcate the difference between minimally to moderately disturbed ecosystem states (Category 2) and highly disturbed ecosystems (Category 3) represent intermediate thresholds that should be subject to more specific considerations because of locally relevant physical, chemical and hydromorphological conditions and management objectives.

The proposed benchmark values in »Table 4.8 concern the oxygen regime, nutrients and Chl-a, pH, temperature, ammonium and the heavy metals aluminium, arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc. The values are based on internationally and nationally established criteria and standards to protect highly intact freshwater ecosystems and to characterize severe ecosystem degradation, respectively. The criteria and standards in the following countries and regional agencies are considered for comparison: Australia/New Zealand, Canada, China, EU, Japan, South Africa, United Nations Economic Commission for Europe (UNECE) and the US. As in the EU WFD only Environmental Quality Standards (EQS) for priority pollutants are established, the numerical values of the UK standards are used for comparison of other indicators. The proposed benchmarks are close to the median values of the criteria and standards in the guidelines considered. See Annex 2 for background information and an overview of criteria and standards which are considered for arriving at the proposed benchmark values.

Table 4.8: Proposed physico-chemical benchmarks for freshwater ecosystems. Annual average total concentrations, unless indicated otherwise

[1] Natural sources and geographical conditions may cause natural background values that differ from the benchmarks for high integrity. Instead of these benchmark values natural background concentrations may be used for setting criteria for high integrity.

[2] Dissolved oxygen concentration varies depending on temperature, pressure and salinity; benchmarks are for freshwater at sea level (760 mm Hg) and 20^{0}C based on the DO%.

[3] Daily average.

[4] Applicable for waters with low hardness (< 60 mg/l $CaCO_3$). In case of higher hardness the benchmark values may be somewhat higher.

[5] Corresponding total ammonia (NH_3 + NH_4^+) concentration depend on pH and temperature. At pH 7.5 and 20^{0}C the benchmarks for total ammonia- N are 1000 µg/l and 6641 µg/l respectively.

	High Integrity (Category I) [1]	**Extreme Impairment (Category 4)**
Dissolved Oxygen Saturation (%)	80 - 120	< 30 or > 150
Dissolved Oxygen Concentration (mg/l)	7.3 - 10.9 [2]	<3 or > 13.6 [2,3]
(optional) BOD_5 (mg/l)	-	>10
Total Phosphorus (TP) (µg/l) - lakes and reservoirs - rivers and streams	 < 10 < 20	 >125 >190
Total Nitrogen (TN) (µg/l) - lakes and reservoirs - rivers and streams	 < 500 < 700	 > 2500 > 2500
Chlorophyll *a* (µg/l) - lakes and reservoirs - rivers and streams	 < 3.0 < 5.0	 > 165 > 125
pH	6.5 – 9.0	< 5
Temperature	No deviation from background value or reference systems or optimum temperature ranges of relevant species	Large deviations from background value or the thermal tolerance range for characteristic species
Un-ionized Ammonia (µg NH_3/l)	15[5]	100[5]
Aluminum (µg/l) pH <6.5 pH >6.5	 5 10	 - 100
Arsenic (µg/l)	10	150
Cadmium (µg/l) [4]	0.08	1.0
Chromium (µg/l) [4] Cr III Cr VI	 10 1	 75 40
Copper (µg/l) [4]	1	2.5

Lead (µg/l) [4]	2	5
Mercury (µg/l) [4]	0.05	1.0
Nickel (µg/l) [4]	20	50
Zinc (µg/l) [4]	8	50

»Table 4.9 summarizes Tier 1 (general screening) and Tier 2 (more detailed screening) requirements for physico-chemical monitoring of the selected most pertinent indicators. This table serves primarily as a checklist. It includes potential indicators for which the analysed guidelines of »Table 4.8 do not provide comparable numerical values.

Tier 1 provides a rapid and overall ecosystem health check. The metabolic indicators provide basic information are water chemistry and light conditions and are readily assessed as part of any field sampling, employing junior expertise. Tier 1 trophic assessment requires easily deployable sampling equipment with basic laboratory analysis, or could be achieved with field kits. The information provides general assessment of nutrient state in the water bodies, but can be suspceptiple to high error if sampling is very infrequent, especially in rivers. N toxicant measures require more sophisticated, or specialised expertise, a laboratory capacity and it is also susceptible to high temporal variability depending on the nature of the water body.

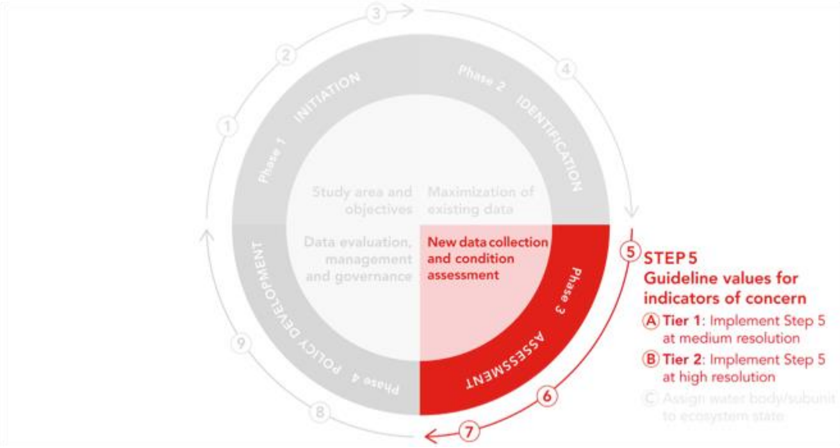
Tier 2 sampling increases the temporal and spatial resolution and, hence, reduces, the potential field sampling error. It dies, however, depend on a greater level of skill and experience by the sampling teams, and judgements (or local knowledge) of where occurrence of pollution, or other impact may be found. This selection is usually related to local knowledge and/or associated with planned interventions close to these monitoring sites.

Table 4.9: Proposed physico-chemical indicators for Tier 1 and Tier 2 level monitoring
Note: Microbial pollutants are listed in this table, because the respective monitoring has already been implemented together with physico-chemical monitoring.

TIER 1					
		Lakes and reservoirs	Wetlands	Rivers (wadeable)	Rivers (non-wadeable)
Indicator group	**Indicator**	**Method**			
Metabolic	Oxygen	DO-spot measurement, minimum and yearly fluctuation			
	Temperature	Annual fluctuations at a few locations			
	pH	Maximum and minimum at some spots			
	Light regime	Secchi depths	N/A	Secchi depths (although sometimes difficult in rivers)	
	Conductivity/ salinity	Conductivity probe	Conductivity probe (in coastal fringes and delta wetlands only)	Conductivity probe	
Trophic	Nutrients N and P	Mean levels of total N and P in agricultural, industrial and urbanized areas			
Toxicants	Heavy metals	Levels of heavy metals (Cd, Hg, Cr, Cu, etc) in agricultural, industrial and urbanized areas			
	Pesticides	Levels of certain specific pesticides in agricultural, industrial and urbanized areas. Selection based on extensive use, environmental fate and toxicity			

	Other organic pollutants	Levels of certain specific organic pollutants like oil, phenol, PCBs in industrial and urbanized areas, potential endocrine disruptors. Selection based on use, environmental fate and toxicity
	Microbial pollutants	Screening for Total coliforms, E. Coli and bacteriophage

Hydromorphological guideline values



Hydromorphological indicators refer to the morphology, batimetry and hydraulics of a river or lake, the discharge regime, and level of suspended matter. As an example of setting hydromorphological guideline the EU WFD guideline for rivers are given in »Table 4.10. For lakes, transitional waters, coastal waters and artificial waters also normative definition are given in the EU WFD. See »Annex 4 for the normative definitions and hydromorphological indicators.

Hydromorphology is an “umbrella discipline” that links hydrology and morphology. Changes in the physical characteristics and processes of freshwater bodies fundamentally affect biodiversity, ecosystem functions, and the related services. Hence, hydromorphological degradation even without corresponding impairment of the physico-chemical status is one of the main reasons of the poor ecological state of many rivers (and other freshwater bodies) worldwide.

Belletti et al. (2014) reviewed 121 hydromorphological assessment methods globally, identifying their main strengths, limitations, and gaps. The fundamental gap identified was insufficient consideration of physical processes.

Table 4.10: Normative definitions of ecological status classification for hydromorphological quality elements of rivers. Source: EC (2000[52]).

Element	High status	Good status	Moderate status
Hydrological regime	The quality, dynamics and the resultant connection to groundwaters, reflect totally, or nearly totally, undisturbed conditions	Conditions consistent with the values specified for the biological elements for good status of rivers	Conditions consistent with the values specified for the biological elements for moderate status of rivers

[52] EC (2000). Water Framework Directive (WFD) 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy, Official Journal of the European Communities L327, 1-72.

River continuity	The continuity of river is not disturbed by anthropogenic activities and allows undisturbed migration of aquatic organisms and sediment transport	Conditions consistent with the values specified for the biological elements for good status of rivers	Conditions consistent with the values specified for the biological elements for moderate status of rivers
Morphological conditions	Channel patterns, width and depth variations, flow velocity, substrate conditions and both the structure and condition of riparian zone correspond totally or nearly totally to undisturbed conditions	Conditions consistent with the values specified for the biological elements for good status of rivers	Conditions consistent with the values specified for the biological elements for moderate status of rivers

A Morphological Quality Index (MQI), developed further during the EU-funded project REFORM[53] covers the full range of physical conditions (e.g., physiographic units, hydrological, climatic conditions) and morphological types of rivers. Main characteristics of the MQI are:

i. it is based on expert judgement,
ii. its application is relatively simple and not too time consuming,
iii. it considers processes rather than just forms,
iv. the temporal component is explicitly accounted for,
v. it considers the hierarchical nature of basins and river networks, and
vi. it allows the deviation from “RCs”.

A detailed description of the entire assessment methods, including case studies, is reported in www.reformrivers.eu. Although rivers and their fringing floodplains are the focus of this assessment method, the key principles can also be transferred to lakes and wetlands.

The first phase of the assessment provides the general setting of the physical conditions and delineates the river network into homogenous reaches, following a 4-steps procedure (Rinaldi et al. 2015; »Table 4.11). The methods applied include remote sensing, topographic, longitudinal and geological maps, GIS techniques, partly field reconnaissance (e.g. Bizzi et al. 2015).

Table 4.11: Summary of the general setting and segmentation procedure.
Source: Modified from Rinaldi et al. (2013).

Steps	**Criteria**	**Outputs**
Step 1: General setting and identification of landscape (or physiographic) units and segments	- Geological and geomorphological characteristics	- Landscape units - Segments
Step 2: Definition of confinement typologies	- Lateral confinement	- Confinement typologies: confined (C) partly confined (PC) unconfined (U)
Step 3: Identification of morphological typologies	- Planimetric characteristics (sinuosity, braiding, and anabranching indices)	- Morphological typologies: Confined: single thread, wandering, braided, anabranching,

[53] www.reformrivers.eu

		Partly confined - unconfined: straight, sinuous, meandering, wandering, braided, anabranching
Step 4: Other elements for reach delineation	- Further discontinuities in hydrology, bed slope, characteristic geomorphic units, bed sediment calibre, channel width, floodplain width	- Reaches

Step 1: General setting and identification of landscape (or physiographic) units and segments. Output: landscape units and river segments.
Step 2: Definition of confinement (i.e. the ratio of channel width to alluvial plain width) typologies. Output: confinement typologies, based on confinement degree and index.
Step 3: Identification of morphological typologies. Output: Morphological typologies, based on sinuosity, braiding and anabranching indicators.
Step 3: Other elements for reach delineation. Output: identification of discontinuities along rivers, based on discontinuities in bed slope, tributaries, dams and weirs, change in confinement and floodplain size, change in sediment caliber.
The second phase of the assessment considers the hydromorphological quality of river reaches:

i. continuity of river processes,
ii. channel morphological conditions, and
iii. vegetation.

Three components are used to assess these quality aspects: geomorphological functionality (forms and processes), artificiality, and channel adjustments. The spatial scale is the river reach, as identified during the first phase. A set of both response and stressor indicators are applied to assess the hydromorphological quality of the reaches.

Finally, the morphological quality index (MQI) can be calculated and assessed against the maximum possible scores that could be reached using the appropriate indicators. The evaluation forms are available in electronic format at http://wiki.reformrivers.eu, allowing the automatic calculation of the indicators once the input values are typed in.

The application of the MQI should be carried out by people with appropriate background knowledge of the underlying principles of fluvial (geo)morphology. The working phases are:

- Collection of existing material
- Preliminary remote sensing – GIS analysis
- Field survey
- Concluding GIS analysis

Table 4.12: Proposed indicators related to hydromorphological indicators for Tier 1 monitoring

Tier 1				
	Lakes and Reservoirs	**Wetlands**	**Rivers (wadeable)**	**Rivers (non-wadeable)**
Indicator group	**Indicator**			
Catchment Land use	% landuse category e.g. agricultural and irrigated, semi-natural grassland, urban, natural or plantation forest			
	% extent and category of protected areas			
	Population density			
	Landscape development index			
	Topography			
Water distribution	Net precipitation			
	Total water abstraction			
	Inter Basin transfers			
Regulation	Net abstraction		eflow regime set	
Connectivity	Impoundment/dam	Drained area	Free-flowing length	

	Permanence of water			
Riparian	Extent shoreline development	Area and % of catchment	Length intact river margins, channelisation	
Thermal regime	Basin volume, hypsograph, seasonal stratification	N/A	N/A	N/A

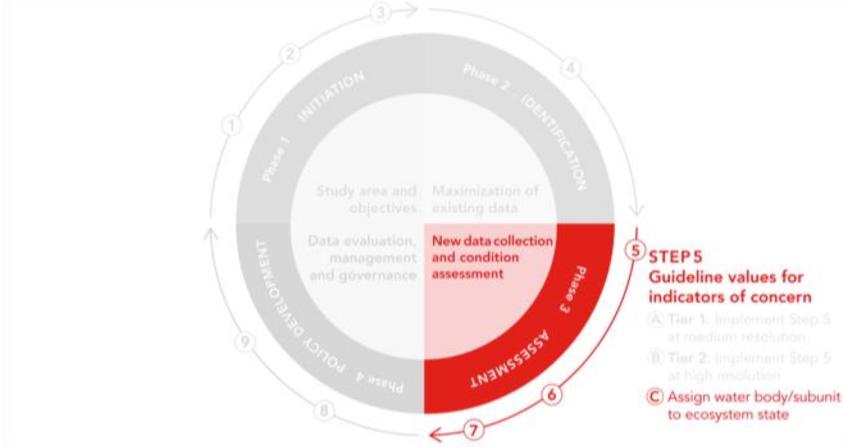
Table 4.13: Proposed hydromorphological indicato s for Tier 2 monitoring of HGS and LGS refers to high and low gradient streams, respectively.

Tier 2				
	Lakes and Reservoirs	**Wetlands**	**Rivers (wadeable)**	**Rivers (non-wadeable)**
Indicator group	**Indicator**			
Catchment Land use	Landscape development index			
	Universal Soil Loss			
	Livestock units			
Geomorphology	Sedimentation rates		% Embeddedness n (HGS), Pool substrate characterization (LGS)	
	Depth of vegetation colonisation	Aquatic vegetation cover and type		Aquatic vegetation cover and type
			Frequency of riffle or bends (HGS), Channel sinuosity (LGS)	
		Use of Wet-Health technique or equivalent	Use of River Hydromorphological Assessment technique or equivalent	
Vegetation colonisation	Depth of vegetation colonisation	Aquatic vegetation cover and type		Aquatic vegetation cover and type
Water distribution	Legally supported water allocation regime			
Connectivity		Inundation regime (Area X Days)	Tributary confluences/Floodplain confluences	
Riparian	Shoreline complexity. Use of Lake Habitat Survey or equivalent		Use of e.g River Habitat Survey or equivalent	

Field surveys should not be carried out during high flow or after major flood events. Field surveys are used to verify remote sensing images and integrate those aspects that cannot be assessed by remote sensing techniques.

Applying these principles to the suggested two-tiered approach for general and more detailed screening respectively is outlined in »Table 4.12 and »Table 4.13. Tier 1 assessment for hydromorphology can be achieved through a desk top exercise, supported where possible with local knowledge. Tier 2, depending on the scale of information requires field assessments by suitably qualified staff.

Assigning water body to a state category (Step 5c)



Ideally and similar to the setting of objectives, the selection of indicators involves a range of relevant and informed stakeholders. These include different levels of the state administration, industry, agriculture, communities, environmental organisations, recreational organisations. This is important as the selection of the indicators is essential to provide adequate metrics to report about the state of water bodies and consequently about success or failure to achieve the objectives. For more detail on indicators and assigning freshwater bodies to certain categories of ecosystem health see »Section 4.5. When measuring agreed indicators and the performance of management actions e.g. in the frame of a monitoring programme - the economic, social and environmental impacts of the management actions - should be evaluated and objectives re-visited (i.e. applying an adaptive management cycle). Monitoring programmes should cover quantitative as well as qualitative characteristics of water bodies. Establishing monitoring objectives, requires substantial effort to designing and executing a programme that is suited to the overall objectives. This involves decisions on e.g. indicators, allocation of resources and organizational infrastructure. Additionally, monitoring is required to evaluate enforcement of regulations and to assist, where applicable, related schemes such as the “polluter pays principle”. More detail on Step 6 i.e. monitoring can be found under »Section 0.

The administration should support these steps in policy making by reporting on issues and technical alternatives and consultation on agendas of stakeholder groups. Channelling back issues between different actor groups can be facilitated setting up multi-stakeholder platforms (Warner, 2007). This feedback towards the scientific community deals with potential water issues and towards the political community deals with potential water agenda items.

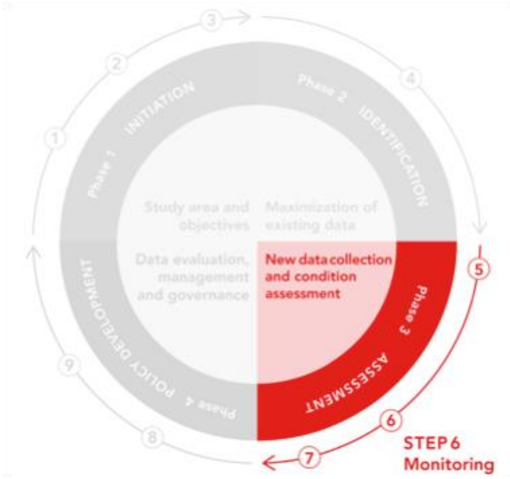
Objectives concerning reporting, public information and stakeholder consultations are important to make transparent how data will be collected and, subsequently, disseminated, and how results relate to pressures, stressors and ecosystem status. The awareness of the important role that stakeholders and citizens can play in the protection of ecosystems is increasing, and requires an adequate strategy concerning data evaluations, reporting and communication. Further, objectives can be formulated concerning the measures to be taken. For example the use of best available techniques (BAT) to prevent pollution, reduction of a certain percentage in pollution discharges into water bodies within a certain period, re-introduction of species, limiting fishing or restricting water withdrawal, etc. could be promoted.

In order to define water quality targets such as the RC or the BAC and to set the thresholds for categories subdividing the freshwater ecosystem quality continuum (»Figure 4.1), the attributes characterizing the actual state of the ecosystem need to be identified as well as indicators and threshold values defined. Furthermore, assessment and monitoring schemes and protocols need to be established for the descriptors of the state of the health of freshwater ecosystems. Corresponding to the four proposed freshwater ecosystem quality categories (»Figure 4.1), a list of indicators and their respective threshold values characterizing the benchmark of Category 1 “High ecosystem status” (dotted bar), the possible demarcations (threshold values) between Categories 2 and 3 as well as the benchmark between Categories 3 and 4 (dashed bar) need to be identified.

The thresholds characterizing potential levels of aspiration should be set based on scientific principles and on the RC of a system, thus the first step is an ecocentric view: what is best and worst state of an ecosystem (i.e. ecosystem integrity and health above the dotted and below the dashed bars), considering its environmental and socio-economic context. The level that a water management authority wants to reach within the middle categories (Categories 2-3) is a societal decision, which brings in the utilitarian view, i.e. the societal value system and reflects the readiness to pay for improvements or ecosystem conservation. The ecological condition gradient method (»Figure 2.13) developed by US EPA (2011b) and presented in »Section 2.7 as well as in »Annex 3 is an excellent albeit data demanding example how to develop a scheme and classify ecosystem health into different categories.

Monitoring, data management and synthesis (Step 6)

N.B. This step matches the 'Monitor' step in the revised Framework (See Preface).



Identification of monitoring objectives

Monitoring is an integral part of the adaptive approach to freshwater ecosystem health protection and rehabilitation. Depending on the particular problems and foci, monitoring may have different objectives. These have been discussed in a general context in »Section 2.6 and as part of the objective setting in »Section 0.

»Box 4.4 summarizes seven key components for successful monitoring modified after to Lovett et al., (2007). The objective setting for monitoring is relevant in all seven aspects of monitoring. »Box 4.5 presents advanced technologies and new approaches in water quality and ecosystem assessment.

Identification of adequate spatial and temporal scales of monitoring and its implementation

Good planning and effective human capacity is essential for cost-effective data collection that supports achieving water quality objectives. Inherent natural variability presents considerable challenges for monitoring, so optimizing the effectiveness of spatial and temporal scales is of fundamental importance. Selection of monitoring locations and the temporal frequency of sampling is guided by the objective of the monitoring. The timing of seasonal minima and maxima of both water chemistry as well as biological communities can be difficult to predict, and assumptions that these patterns are similar even in the same region can be quite misleading (Irvine 2004). Pressures, stressors and impact assessment identify what should be monitored.Attention to the seasonal dynamics of water bodies guides when and where samples should be collected. The majority of sediment and phosphorus loading for example can be driven by a small number of high rainfall events, easily missed by routine "grab" samples. In some situations it is possible to sample for water chemistry continuously, but this is only possible where there is sophisticated and well maintained infrastructure. In most parts of the world this is not the case, necessitating sound judgements based on best available and local knowledge. Archival information can be highly useful in selecting monitoring locations.

But, even where previous measurements are absent, remote sensing data can not only guide likely locations for effective monitoring, but provide useful data for some variables or parameters. In river networks, monitoring can be targeted to both where land use and human settlements are most intensive, and where they are not, to obtain representative sites with "best" and "worse" quality status. Confluence of tributaries also guide sample locations and sampling below or above these points, or both, is guided by overall objectives and resources. Monitoring regimes are often tiered, ranging from occasional monitoring as a "health check or screening" to intensive monitoring to locate and address sources of pollution. In the EU WFD,

(1) Design the programme around clear and compelling objectives and on scientific principles. Objectives are crucial because they determine the variables measured, the spatial extent of sampling, the intensity and duration of the measurements, and, ultimately, the usefulness of the data.
(2) Include review, feedback, and adaptation in the design. The objectives may change over time, and any programme should have the capacity to adapt to changing objectives and incorporate changing technology without losing the continuity of its core measurements.
(3) Choose measurements carefully and with the future in mind. The core of monitoring should be defined to capture basic measures of the functioning of the system, indicators of change, and variables of particular importance for ecosystems. Measurements should be as inexpensive as possible because the cost of the programme may determine its long-term sustainability.
(4) Maintain quality and consistency of the data. The best way to ensure that data will be used is to avoid that quality is compromised and measurement methods or collection sites maintained. The confidence of future users of the data will depend entirely on the quality assurance programme implemented from the beginning. Sample collections and measurements should be rigorous, repeatable, well documented, and employ accepted methods. Methods should be changed only with great caution, and any changes should be recorded and accompanied by an extended period in which both the new and the old methods are used in parallel, to establish comparability.
(5) Plan for long-term data accessibility and sample archiving. Metadata should provide all the relevant details of collection, analysis, and data reduction. Raw data should be stored in an accessible form to allow new summaries or analyses if necessary. Raw data, metadata, and descriptions of procedures should be stored in multiple locations. Data collected with public funding should be made available promptly to the public. Policies of confidentiality, data ownership, and data hold-back times should be established from the beginning. Archiving of soils, sediments, plant and animal material, and water and air samples provides an invaluable opportunity for re-analysis of these samples in the future.
(6) Continuously examine, interpret, and present the monitoring data. The best way to catch errors or notice trends is to use the data rigorously and often. Adequate resources should be committed to managing data and evaluating, interpreting, and publishing results.
(7) Include monitoring within an integrated (research) programme. An integrated programme may include modelling, experimentation, and cross-site comparisons. This multi-faceted approach is the best way to ensure that the data are useful and, indeed, are used.

(Adapted from Lovett et al, 2007).

Box 4.4: The seven habits of highly effective monitoring programmes

Although the selection of the location is highly dependent on the objectives, some general rules exist such as:

- river health monitoring should cover sites across a gradient from headwaters to coastal zone:
- for water chemistry analysis samples should be collected midstream; and
- river invertebrates are usually sampled from the most biodiverse biotopes/habitats (often riffle zones; Kelly et al. 2008).

In lakes, open water samples for water chemistry and lake structure defined by temperature and oxygen stratification of the water column have been used effectively to assess overall condition. Inshore samples provide information on localised stresses, but results can be influenced by littoral habitat structure and prevailing wind.
In order to assess general species richness e.g. related to questions of biodiversity, sampling of multiple habitats can be effective, but if impacts from identified stressors are targeted, habitat-specific sampling is preferred since it reduces inherent variation (see »Section 2.5.6 and »Section 4.6.3). Otherwise targeting a specific habitat (consistently and based on its commonness) reduces inherent variation and enhances the identification of response to a particular stressor.

In wetlands hydrogeomorphic units defined by a mix of vegetation structure and water regime form the basis of monitoring using protocols such as defined by Wet-Health (http://www.wrc.org) and Maltby (2009). The Wet-Health techniques are gaining in use and assess current condition of a wetland against a theoretical or expert-judgement based reference condition.

The future monitoring and assessment of freshwater ecosystems and water quality will greatly benefit from rapidly developing technologies, involvement of citizen, and novel approaches (e.g. Newman et al. 2012, Krause et al. 2015). In particular citizen-science activities, interlinked with professional scientists and managers, have the potential to fundamentally change how data are collected – following standardized protocols, how data are analyzed and visualized, and how information is communicated. Both local and large-scale issues may be addressed, which is particularly important in areas that are not yet covered by routine monitoring and assessment programs. For example, mobile applications (apps) and wireless sensor networks, which connect laboratories and information systems with the natural environment, show great promise for advancing ecosystem assessment.

Remote controlled systems including light unmanned aerial vehicles (UAV) offer new opportunities to measure and monitor ecological processes at relevant spatiotemporal scales, which are defined by the user. The rapid development of miniaturized sensors attached to UAVs will further facilitate ecosystem assessment and monitoring (Anderson & Gaston 2013).

Similarly, the recent development and application of advanced technologies from the generically defined “-omics” sciences (e.g. genomics) coupled with bioinformatics platforms provide new and advanced insights into understanding biodiversity patterns and changes; but also of harmful algae blooms developments, which have increased over the past few decades and affect public health and ecosystems alike (e.g. Anderson et al. 2012).

Box 4.5: Water quality and ecosystem assessment: Advanced technology and new approaches

Although the selection of the location is highly dependent on the objectives, some general rules exist such as:

- river health monitoring should cover sites across a gradient from headwaters to coastal zone:
- for water chemistry analysis samples should be collected midstream; and
- river invertebrates are usually sampled from the most biodiverse biotopes/habitats (often riffle zones; Kelly et al. 2008).

In lakes, open water samples for water chemistry and lake structure defined by temperature and oxygen stratification of the water column have been used effectively to assess overall condition. Inshore samples provide information on localised stresses, but results can be influenced by littoral habitat structure and prevailing wind.

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In wetlands hydrogeomorphic units defined by a mix of vegetation structure and water regime form the basis of monitoring using protocols such as defined by Wet-Health (http://www.wrc.org) and Maltby (2009). The Wet-Health techniques are gaining in use and assess current condition of a wetland against a theoretical or expert-judgement based reference condition.

Definition of sampling frequencies for the different indicators

Different physical, chemical and biotic indicators provide complementary information on water quality for ecosystems. Monitoring programmes need to be responsive to local hydrology, climate and landscape (Vos et al. 2000). Statistically robust sampling requires an understanding of the effect of sample and site variation (Stoffels et al. 2005) to guide sample frequency and intensity, but statistical confidence is affected by sample size (including sub-sampling procedures), within site replication, habitat sampled, taxonomic resolution and statistical treatment of data. Recognising the influence of local physical conditions allows flexibility in design so that monitoring provides the information necessary to assess the overall quality to guide assessment and management. However, statistical techniques as well as being powerful aids to monitoring protocols and data analysis can have a low predictive ability and pose a high risk of misclassification (Håkanson 1999), whereby high heterogeneity requires more samples. Decisions on what to measure and its resolution naturally affect costs. Generally, biological samples are less prone to short term fluctuations of environment than chemical ones (Resh and Jackson 1993), but collecting biological samples is usually a fraction of the costs involved in sorting and processing them for identification. Taxonomic sorting of samples to species level is much more costly than to genus or family, but as a consequence there is some loss of information. Collecting biological samples from restricted habitats reduces inherent statistical noise and may be a preferred choice when monitoring is designed to link a particular stressor to an impact (Johnson et al., 2004; Pinel-Alloul et al., 2006). However, the optimal strategy for sampling depends on the purpose of monitoring and the importance of the interaction of multiple environmental gradients. Pooling samples across individual sites can be an effective and cost efficient approach that amalgamates information across a water body (Snell and Irvine 2012).

Biological indicators are, nevertheless, prone to the rhythm of the season including biological migrations which is especially relevant for some fish species. The frequency of sampling for all indicators is based on the cost-effective trade-off between spatial and temporal coverage and the overall intention for longevity of monitoring. Some national programmes for monitoring rivers sample every few years to detect long term trends. However, in the early stages of a sampling programme, more intensive sampling is recommended to guide monitoring effectively. The cost effectiveness of long-term monitoring even at low intensity is well established and coherently argued (Lovett et al., 2007). While increasing the number of biological samples across different times of the year increases likelihood to detect change, for biological sampling, a diminishing return on effort is common beyond two or three sampling occasions (Blocksom et al., 2002; Halse et al., 2002). For all water bodies, manifestation of pressures and stresses at the local scale are nested effects within larger scale effects. When employing biological sampling, especially at the community level, there is contrasting evidence and opinion on the optimum taxonomic resolution required for biological assessment (Hawkins et al. 2000; Bailey et al. 2001). Some authors argue strongly for identification of taxa to the lowest possible taxonomic level (Blocksom et al., 2002), namely species (e.g. Furse et al., 1984) or genus (Yoder and Rankin 1995). Others claim that higher taxonomic resolution provides sufficient information, but at much reduced financial costs. This debate is complicated further by e.g. the differing treatments of rare taxa, the use of mixed taxonomic resolution and by the process of sub-sampling. However, most taxa are rare, occurring at few sites at high densities or many sites at low densities. Therefore, either distribution range or abundance patterns can be used to assess rarity (Nijboer and Verdonschot 2004).

Consequently, rare species or taxa are often removed from data sets before analysis, or are “downweighted” (e.g. Heino 2000; Bailey et al., 2004; Johnson et al., 2004). For routine monitoring for water quality Hämäläinen et al. (2003) advise not to place much emphasis on rare taxa that tend to wax and wane for natural reasons or because of unrepresentative sampling. However, low impacted sites tend to have greater taxa richness, and more rare species, than impacted ones (Doberstein et al., 2000; Fairchild et al. 2000; Chase and Liebold 2002), such that rare species may be critically important indicators of ecosystem health (Lyons et al. 1995; Cao et al. 1998). These discussions have important implications for sampling strategies (Doberstein et al. 2000) and emphasis the necessity for clarity as to the purpose of a monitoring programme (including baseline assessments). For example, in many cases, family level or morphotype-based information has been shown to suffice, e.g. for rapid bioassessments using benthic macroinvertebrate communities, as part of a rapid or intermediate level environmental flow determination in places with limited resources or published taxonomic guidance. This is reflected in the recommendations for Tier 1 and Tier 2 level monitoring (and Table 4.7).

Data management, quality assessment and control

Good monitoring includes provisions for management, human capacity including training needs, and accessibility of data (Lovett et al. 2007). Management protocols guide sampling procedures and data management. All sampling should, therefore, be supported with Standard Operating Procedures, and guided where possible by published national and international standards. Examples are provided by the Comité Européen de Normalisation (CEN) standards, and include those published by British Standards (BS) and the European Norm (EN): BS EN 14184:2014 on Water quality-Guidance standard for the surveying of aquatic macrophytes in running waters; BS EN 14996:2006 on Water quality-Guidance on assuring the quality of biological and ecological assessments of the aquatic environment; BS EN 14962: 2006 on Water quality-Guidance on the scope and selection of fish sampling methods; BS EN 14614:2004 Water quality-Guidance standard for assessing the hydromorphological features of rivers; BS EN 16665:2005 on Water quality-Guidance for quantitative sampling and sample processing of marine and soft-bottom macrofauna; EN 13946: 2014 Water quality-Guidance standard for the routine sampling and preparation of benthic diatoms from rivers and lakes; and EN 144407: 2014 Water quality-Guidance standard for the identification, enumeration and interpretation of benthic diatom samples from rivers and lakes. The standards provided above serve as examples. These standards and similar others provide an outline of good practice and quality assurance.

ISO/IEC 17025:2005 specifies the general requirements for the competence to carry out tests and/or calibrations, including sampling. It covers testing and calibration performed using standard methods, non-standard methods, and laboratory-developed methods. It is applicable to all organizations performing tests and/or calibrations.

The ISO/IEC 17025 standard itself comprises five elements that are Scope, Normative References, Terms and Definitions, Management Requirements and Technical Requirements. The two main sections in ISO/IEC 17025 are Management Requirements and Technical Requirements. Management requirements are primarily related to the operation and effectiveness of the quality management system within the laboratory. Technical requirements include factors which determine the correctness and reliability of the tests and calibrations performed in laboratory.

Laboratories use ISO/IEC 17025 to implement a quality system aimed at improving their ability to consistently produce valid results. It is also the basis for accreditation from an accreditation body. Since the standard is about competence, accreditation is simply formal recognition of a demonstration of that competence. A prerequisite for a laboratory to become accredited is to have a documented quality management system. Other standards can be accessed at ISO websites[54].

Data archiving, metadata and storage

Simple to use data storage and retrieval systems are required in order to compare monitoring results, and access associated reports and background information. Metadata provides information about the data, and procedures for this are supported by e.g. ISO 19115-1:2014 [55]and associated metadata standards. This is essential in order to put data in a proper context, and to document attributes of the collection process. Metadata is specific for each data set, and should provide information on how to access sampling rational and protocols, standardisation of variables, taxonomy, station codes, geographic information, information on data sources, data analysis and summarising, and data ownership. Information on standard operating procedures should be kept updated and easily accessible. Raw data should be stored in an accessible form. Archiving of samples where this is possible should be well documented. Archiving biological samples is particularly important where these may need to be checked in the future. This includes professional storage and, for sampling dependent on species taxonomy, documenting of type-specimens.

A basic philosophy supporting open access to data provides the best means for enabling water resources management. Information sharing systems are required across agencies and, for transboundary waters, among neighbouring political jurisdictions. Where there is reluctance for data sharing, or an incompatibility of both collection methods and electronic access to data across agencies, this greatly hinders effective governance of water bodies and their ecological assessment. Well documented procedures, rational and quality assurance adds confidence to the legitimacy of reported results. Well managed data storage does not only support knowledge and promotes good governance, it also helps ensure timely response to ecosystem degradation and improve data safeguard security owing to clearly defined procedures that underpin data access. Transparency of data collection methods and access of monitoring results improves stakeholder confidence and the legitimacy of monitoring. Shared access to data linked to GIS is increasingly feasible through development of reliable open-source computer programmes and networks. Real and perceived risks of data sharing (UNECA et al., 2003) are generally readily overcome through sound and collectively developed data sharing policies.

Accessing data requires effective data management systems and Spatial Data Infrastucture (SDI) linking GIS to data management systems. Sharing data requires clearly defined and collectively agreed policies among data providers. Developing procedures for sharing GIS based data can be guided by the SDI Cookbook (GSDI 2013), the SDI for Africa Handbook (UNECA et al., 2003) and the United States Geological Survey data policy documents (USGS 2013). Quality assurance of data provision and use should also address issues of liability and accuracy. In general the benefits of sharing data greatly outweigh the risks, and any real of perceived negative consequences can be addressed in well formulated policies (UNECA et

[54] http://www.iso.org/iso/home.html
http://standardsdevelopment.bsigroup.com/Home/Committee/50002180?type=m&field=Status
[55] http://www.iso.org/iso/home/store/catalogue_ics/catalogue_detail_ics.htm?csnumber=53798

al., 2003). However, effective policies need to be complemented by effective management, and appropriately skilled data managers. This is a component of institutional setting and capacity.

Data synthesis and scoring

Effective synthesising and summarising of collected data is a fundamental requirement for assessing and communicating water quality information. Spatially distributed data is usually aggregated to reflect average conditions within sites or overall water bodies, and accompanied by appropriate estimates of uncertainty. For basic monitoring and reporting the way data are synthesised is determined by the agreed sampling methods, so that there is clarity of what the results are intended to represent. This can vary depending on inherent variability of habitats sampled (e.g. Solimini and Sandin, 2012). To capture meaningful averages it is likely that more biotic samples are collected and processed than those for water chemistry. To capture average conditions at the scale of the water body, pooling of samples and data is often necessary. Resource constraints often lead to the samples being pooled, with subsequent sub-sampling to provide for an average sample, but this looses information on uncertainty among sites.

Synthesis of temporally distributed monitoring results ideally involves some type of time-series analysis, in order to detect long-term trends in data. Seasonal variation of variables complicates annual synthesis, but can be accommodated in time-series analysis. However, as implied by the topic, time-series requires long runs of data to enable a detection of change. The value of long-term monitoring is well advocated (Lovett et al., 2007; Sandin and Johnson, 2000) whereas often, and unfortunately, disregarded.

Measuring key chemical variables, such as toxic substances and nutrients, has value for water quality assessment but is prone to seasonal and, sometimes, daily variation. Chemical water quality assessments only provide an indirect assessment of ecological condition. However, using individual biological metrics to detect water quality, akin to nineteenth century mineworkers using a small bird (the canary) to detect toxic air, is an ideal, but generally an unrealistic aspiration for assessing ecosystem quality. Ecosystems comprise complex and interacting biological and abiotic matrices, therefore inclusion of single or limited taxa groups may only be useful for specific aspects of the environment and over limited scales (Allen et al., 1999; Nijboer et al., 2005). The logic that using a number of indicators provides a better overall view of the ecological condition of a water body is the basis of a multi-indicator and multimetric approach to monitoring and assessment (Karr and Chu 1999; Barbour and Yoder 2000; Blocksom et al., 2002; Gabriels et al., 2010; Miler et al., 2013). A biometric approach aggregates a number of metrics into a single quality score. While the use of multimetric assessment of water bodies is increasingly advocated for use in national monitoring programmes (Hering et al., 2006) and underpins the philosophy for ecological assessment under the EU WFD (EC, 2000), it is also prone to over-emphasizing the importance of some individual (and perhaps unreliable) metrics and compounding statistical error with increasing number of metrics. A parsimonious approach is required, whereby care is taken to use a valid minimum number of meaningful metrics, rather than the assuming that more is better, since the latter can lead to diminishing rather than improving robust assessment.

Biometrics fall within a number of categories (Dodkins and Rippey, 2007):

1. Direct ecological response metrics that assume that some emergent properties of an ecosystem reflect quality without being calibrated against a particular stressor. Examples include metrics of species diversity and biomass. Such metrics are inherently prone to misinterpretation.
2. Expert judgment of a perceived ecological change, based in individual or a set of clues and experiential knowledge. While lacking rigor and subject to individual bias, they can also capture in-depth knowledge of particular sites. Application across sites with less familiarity can be inherently unreliable.
3. Taxa metrics calibrated against an impact gradient, using mathematical techniques and scaled against a stressor using either individual taxa or a combination of taxa

into a multimetric score.

4. Measures of the functional characteristic of an ecosystem, such as energy flow, species traits and predator-prey dynamics.

The most commonly used metrics are within the above listed categories 2 and 3. In assessing response of a taxon to an impact, species optima to prevailing conditions are often determined to provide *a weighted average* (Kent and Coker, 1992; ter Braak and Looman, 1986; Denys, 2004), and requires knowledge of pressure and species abundance at several points along a gradient of pressure:

$$S_j = \frac{\sum_{i=1}^{n} A_{ij} W_i}{\sum_{i=1}^{n} A_{ij}}$$

where	S_j	= the score (or optima) for species j
	$\sum_{i=1}^{n}$	= the summation across all the sites where the species occurs
	n	= number of sites
	A_{ij}	= abundance of species j at site i
	W_i	= concentration of the pollutant at site i

As with many techniques that relate a stress to an impact, field results are subject to natural variation, and final relationships subject to inherent variability and uncertainty. This can be reduced by extensive sampling in developing the metric.

Principles in developing appropriate metrics and their amalgamation into multimetrics are outlined in e.g. Borja and Dauer (2008), Breine et al. (2007), Carstensen (2007), Hering et al. (2006), Herlihy et al. (2008), Pont et al. (2006, 2009) and Stoddard et al. (2008) and summarised into a step by step process by Hering et al. (2012[56]).

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Combination of scores, weighting and evaluation of monitoring results

It is often desirable to simplify the detailed and complex information from multiple indices into a single score. This requires determining an appropriate balance between effective communication (simplifying the results for a broader audience) and the risks of losing valuable information by combining or integrating indices. The benchmark values for ‘reference’ and ‘fail’ can be rescaled for each indicator (e.g. from 1 to 0) so that all indicators are comparable.

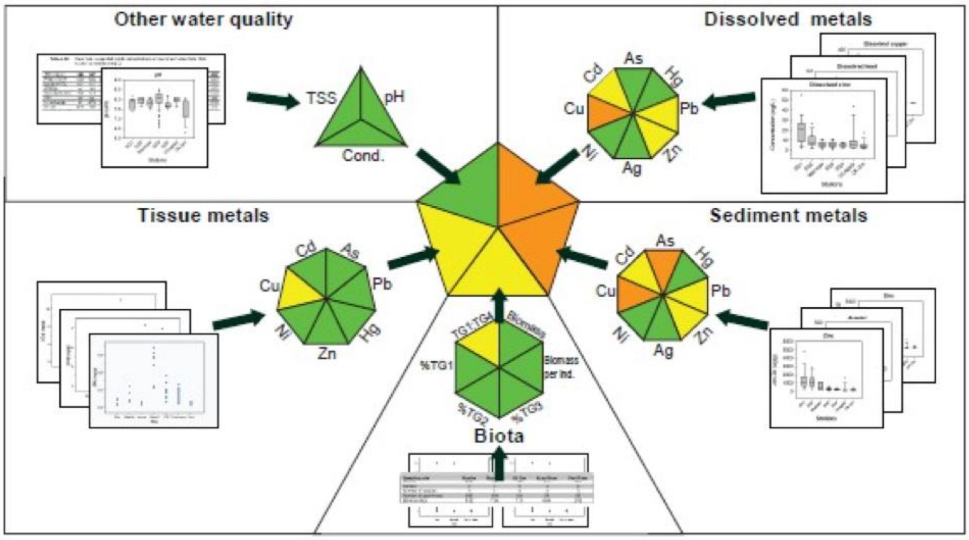
Observed values can be then normalized. Individual indices can then be combined within an indicator group (e.g. indices relating to fish can be combined as a single ‘fish’ score. Several water quality parameters can be grouped as an overall water quality indicator as shown in »Figure 4.7.

When combining indices, it may be necessary to take the minimum score rather than the

[56] http://www.wiser.eu/download/D2.2-2.pdf

average. For example, if the water is toxic for one heavy metal, then it should be regarded as toxic even if all other water quality indices do not fall below the benchmark or threshold of concern value (see »Figure 4.7). When combining indicator groups into a single score, different weightings may be given to some indicator groups depending on the specific objectives goals of the programme. For example, a higher weighting might be given to biological indicators if the primary objective is for biodiversity protection.

Figure 4.7: Complex data sets
Water quality of the Strickland River in Papua New Guinea. Each water quality indicator was re-scaled (1 = reference to 0 = fail) based on benchmark values. Combined indicator groups are colour-coded: as acceptable condition (green), exceeding a threshold of concern (yellow) or failing (orange).
Source: http://www.barrick.com/files/porgera/PEAK-Porgera-Report-Card-2010.pdf



Evaluation, quality category assignment, reporting and communication (Step 7)

N.B. This step matches the ‘Evaluate and report’ step in the revised Framework (See Preface).

An essential element of monitoring and assessment is to inform those responsible for policy and management so they can respond and address emerging issues, either as a technical adjustment of the actual policy or as start of a new policy cycle. Reasons for “technical adjustment" could be that the indicators chosen are shown to be unreliable or practicably difficult to apply. Reasons for the start of a new policy cycle could be that wishes and demands of the society changed over time (political agendas shaped by interest groups) or new or different objectives identified by research. Reporting involves the effective communication of key findings of monitoring and assessment, and is an essential process in closing the adaptive management loop. Technical reports are often used to convey important environmental information to a specialist audience (von Schiller et al., 2001). However, they are not particularly effective at reaching decision makers and other stakeholders, since the language is often complex and the publications are not easily accessible. It is often desirable to communicate findings more broadly to engage interest and support from the public for management intervention. This often necessitates particular communication skills and associated capacity development within the relevant institutions.

Key questions to consider are e.g. 'who is the report being prepared for' and 'for what purpose'? Different audiences (e.g. scientists, policy makers, general public) may require different levels of detail and different strategies for communication and engagement. Environmental report cards (examples shown in »Figure 4.8 and »Figure 4.9) have emerged as an effective tool to integrate monitoring data, provide feedback for a wide audience, and importantly, close the adaptive management loop (see »Section4.8.1). They can be used to raise environmental awareness and engage citizen scientists, but also catalyse management actions and track their effectiveness. These can take many forms, depending on the target audience, and should be based on principles of effective science communication (Dennison et al., 2007). Examples include the South African River Health Programme (RHP) (»Box 4.6), US Ecological Monitoring and Assessment Programme (Jackson and Paulsen 2009), and the Ecosystem Health Monitoring Programme in Southeast Queensland, Australia[57] (Bunn et al. 2010). Similar approaches have recently been applied in China[58].

Additionally to reporting on the condition of individual sites at a particular point in time, it may be valuable to report on the proportion of sites in a region that are passing or failing, or to report on trends over time. Are most sites in good/bad condition or improving/degrading over time? The way in which information is presented in a report card must make sense to the audience. It is primarily a communication tool and needs to be underpinned by a more technical report on the data[59].

The freshwater ecosystem health monitoring programme uses a similar A-F reporting system to that used in school reports. Source: www.ehmp.org.

[57] www.ehmp.org
[58] www.watercentre.org/research/rhef/project-resources/report-cards/reportcards
[59] For an example, see www.ehmp.org

Figure 3.4.2: Report cards on ecosystem health in Southeast Queensland, Australia. The freshwater ecosystem health monitoring programme uses a similar A-F reporting system to that used in school reports. Source: www.ehmp.org.

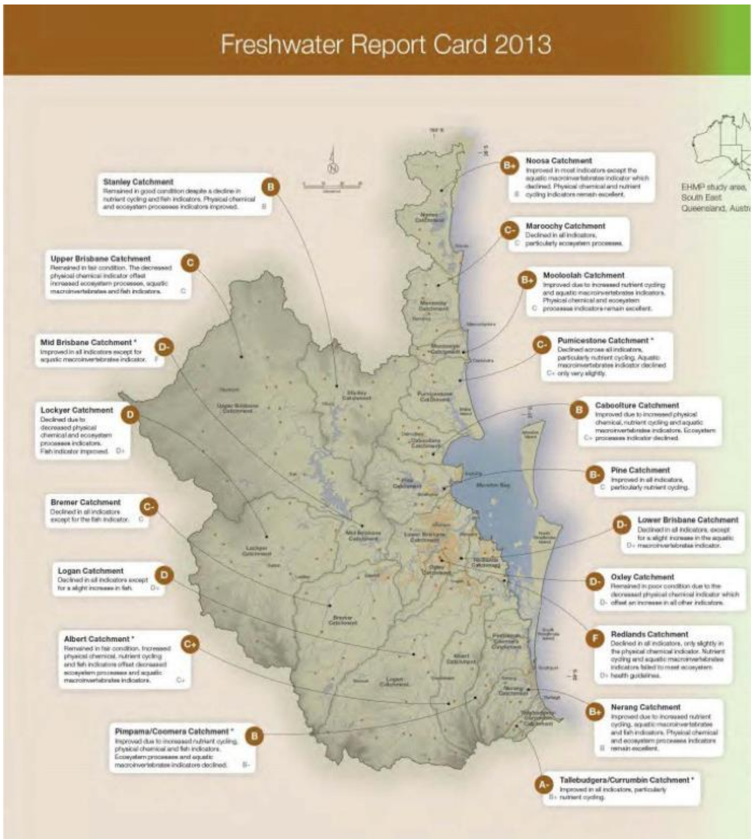
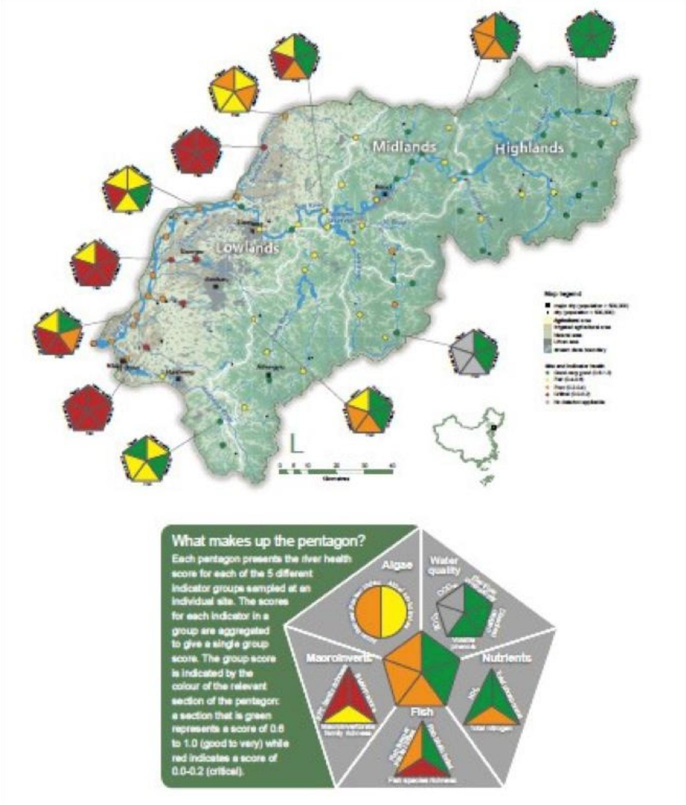


Figure 4.9: Report card on ecosystem health in the Taizi River in China uses a similar suite of indicators to those of Southeast Queensland but a colour-coded presentation.
Source: www.watercentre.org/research/rhef/project-resources/report-cards/reportcards



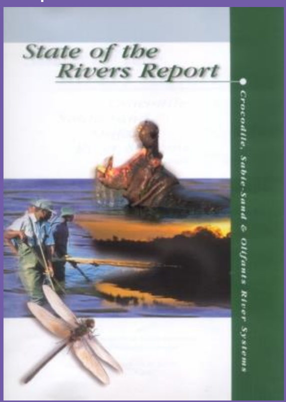
The South African RHP was initiated by the Department of Water Affairs and Forestry (DWAF) in 1994, to serve as a source of information regarding the overall ecological status of river ecosystems in the country. The objectives of the RHP (DWA 2011) are to:

- Measure, assess and report on the ecological state of aquatic ecosystems
- Detect and report on spatial and temporal trends in the ecological state of aquatic ecosystems
- Identify and report on emerging problems regarding aquatic ecosystems
- Ensure that all reports provide scientifically and managerially relevant information for national aquatic ecosystem management

The RHP primarily utilizes instream and riparian biological communities (e.g., fish, invertebrates, vegetation) to characterize the response of the aquatic environment to multiple disturbances. The rationale is that the integrity or health of the biota inhabiting the ecosystem provides a direct and integrated measure of the health of the river as a whole. Implementation of the RHP has largely been through provincial teams comprising, among others, DWA Regional Offices, provincial departments of the environment, conservation agencies, universities, and municipalities. Implementation has largely been voluntary. It remains vulnerable to and is influenced by factors such as the degree of enthusiasm of provincial champions and task teams, buy-in from their respective organisations, and the availability of financial and human resources.

State of the rivers (SoR) reports have been compiled over the past decades for many of the South African river systems through the RHP, and a supporting Rivers Database has been set up for the collation of biomonitoring data. Examples of these various reports and supporting posters, used as tools for teaching purposes and for communicating to civil society the state of health of different river systems at different scales (e.g. individual river basins, entire Water Management Areas, major metropolitan areas), as well as a SoR reporting manual, can be downloaded from https://www.dwaf.gov.za/iwqs/rhp/state_of_rivers.html).

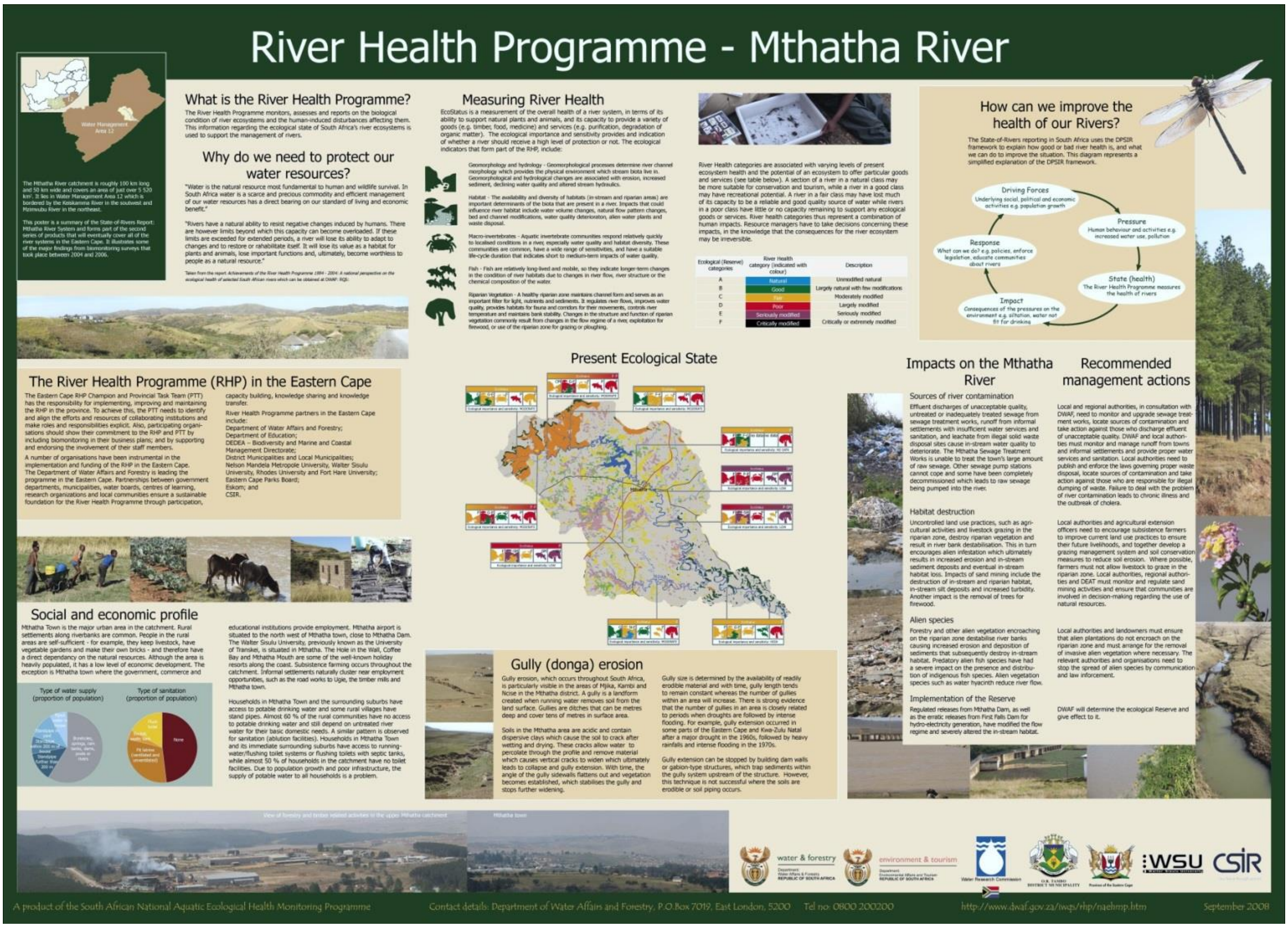
Examples of materials include:
(1) A state-of-the-rivers report for the Crocodile, Sabie-Sand and Olifants River Systems (2001). Source: http://www.dwaf.gov.za/iwqs/rhp/state_of_rivers/crocsabieolif_01_toc.html)
http://www.watercentre.org/research/rhef/project-resources/report-cards/reportcards



(2) A poster synthesizing the RHP results for the Mthatha River System, Eastern Cape (2008), which is displayed in Figure 4.10.
Source: https://www.dwaf.gov.za/iwqs/rhp/state_of_rivers/posters/Mthatha-6-e.pdfhttp://www.watercentre.org/research/rhef/project-resources/report-cards/reportcardshttp://www.watercentre.org/research/rhef/project-resources/report-cards/reportcards

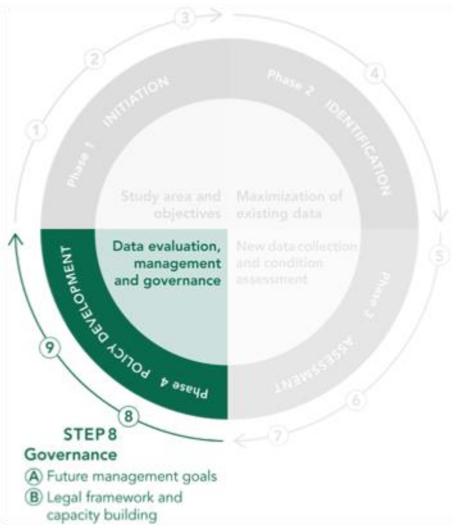
Box 4.6: The South African River Health Programme

Figure 4.10: River Health Programme (RHP) results (for the year 2006) for the Mthatha River System, Eastern Cape.
Source: https://www.dwaf.gov.za/iwqs/rhp/state_of_rivers/posters/Mthatha-6-e.pdf



Governance, legal frameworks, adaptive management and capacity issues (Step 8)

N.B. The issues discussed in this step are mainly addressed in the 'Assess capacity' step, and the 'governance band' in the revised Framework (See Preface).

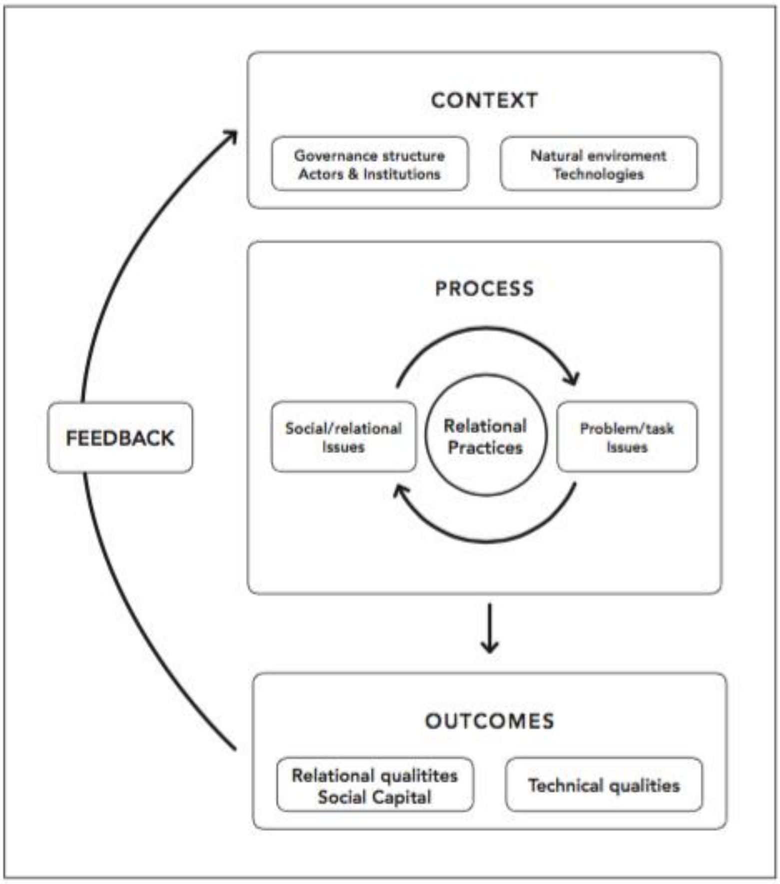


Governance legal frameworks and adaptive management (Step 8a)

As discussed in general in »Section 2.9, the management of water resources, including water quality and quantity, is an adaptive management task which requires its corresponding, enabling governance framework. The emphasis on adaptive management does not contradict the concept of IWRM. To the contrary, it is a requirement to actually realize IWRM given all the uncertainties water management has to deal with. Adaptive management emphasizes the consideration of uncertainties (both stochastic and epistemological) inherent in many tasks related to water resources management. Given the complexity of tackling water management from an integrated perspective, it is essential to take such uncertainties into account and to adopt a more adaptive and learning management approach. The no, or low regret steps are to be taken first and continuous monitoring, assessment and (re)evaluation of the water system performance should help avoid undesirable developments by modifying objectives and approaches in due course. Adaptive management models allow the consideration of evolution in objectives, aspiration (target) levels and in the value system. The adaptive management model builds on a social learning framework which links the context, the process of analysis, negotiation and decision making and the outcome in a unique, repetitive feedback learning loop as displayed in »Figure 4.11.

In the centre of the societal learning framework are processes of learning and negotiation that are influenced by the context in which they are embedded. These processes produce outcomes that may lead to changes in the context, and thus to a cyclic and iterative long-term process of change (Pahl-Wostl et al. 2007). This closed loop feature allows entry into the process at any stage. However, recognizing the influence of context leads to the conclusion that it is essential that an adequate governance structure is in place. This involves both formal and informal elements and the adequately trained professional capacity to implement, and preferably public involvement to support this. Social learning can improve context but it will not take place without the appropriate enabling environment.

Figure 4.11: Conceptual framework for social learning in resources management
Source: Pahl-Wostl et al. (2008).



»Figure 4.11 also highlights that solving what is perceived as a technical problem (e.g. developing an innovative monitoring programme for water quality) cannot be detached from social/relational issues (e.g. how are the problem and the task framed, who is involved, how is political power distributed). In particular adaptive management approaches require the respect of the principles of good governance. It must be transparent who decides on what kind of evidence, if and how management objectives or measures should be modified. Otherwise flexibility and the possibility to modify management objectives and measures may be used by some groups to impose their vested interests on the management process. The adaptive management/governance concept, as reflected also in the 4 phases - 9-step-approach as described in »Sections 4.1 - 0 clearly underlines that every technical, scientific task (like water quality management and restoration/safeguarding of freshwater ecosystem health) is inherently and simultaneously a societal decision making and negotiation process. Hence, the system analysis of such processes should not separate the societal and ecological dimensions.Because the skills and experience may differ between the trained social and natural scientists particular attention is needed to develop a shared sense of purpose and communication across professionals involved in the different stages of monitoring and management.

This interlinkage between the societal and ecohydrological dimensions is well documented in the 'Sustainable Management of Hydrological Alterations' (SUMHA) Framework (Pahl-Wostl et al. 2013). This framework builds upon a recent combined scientific and social process developed to determine environmental flows for different types of river system at the landscape scale of an entire region, state or large basin, the Ecological Limits of Hydrologic Alteration (ELOHA) (Poff et al. 2010). The application of ELOHA (»Box 2.2) allows the more rapid scaling up of efforts to set flows to maintain ecosystem health beyond individual systems or projects. The additional components in SUMHA provide for more explicit integration of the Governance and Management System (GMS) of influence and of tradeoffs in ES and system health.

Figure 4.12: Scheme for a sustainable and adaptive management of water quality derived from the Sustainable Management of Hydrological Alterations (SUMHA) Framework. GMS refers to the Governance and Management System.
Source: Pahl-Wostl et al. (2013).

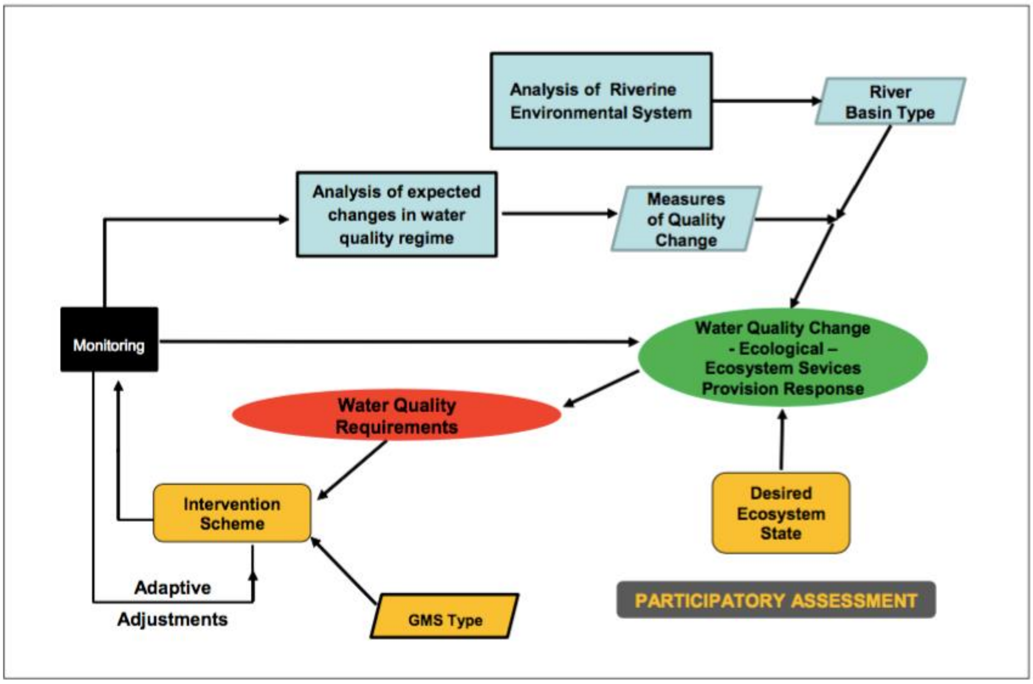


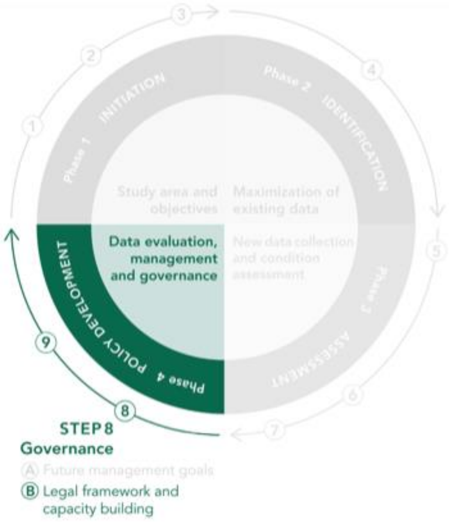
Figure 4.12. This approach highlights freshwater ecosystem health as a joint quantitative and qualitative water management task (blue and ochre boxes in »Figure 4.12). In this problem-solving flow chart the water quality requirements of freshwater ecosystems (red oval in

»Figure 4.12) links the ‘natural’ subsystem with the ‘societal ‘one. The societal process has the goal to find a compromise solution which is acceptable for society and feasible according to the natural resources and ecosystem functions. This requires an iterative process where multi-stakeholder negotiations and trade-off analyses take place repeatedly. Negotiation processes are linked to (experimental) interventions and observation (monitoring) of the hydro ecological system. The framework highlights that development and implementation of interventions need to be tailored to the specific characteristics of the GMS.

»Figure 4.12 indicates the crucial role of monitoring for all three sub-domains: societal decision finding, hydrological and ecological processes. This scheme moreover underlines the importance of comprehensive and long term monitoring as the pre-requisite of – not only adaptive but – all types of natural resource and ecological management by following the “What you do not measure you cannot manage” principle. Next to the cyclical (loop) characteristics of »Figure 4.12 the relatively closed nature of the process is obvious. Should the iterative decision making/monitoring process fail to achieve a satisfactory outcome (solution) within the given GMS, the feedback to the higher order (political/legislative) system (not shown in »Figure 4.12) is due to modify/develop the GMS. This implies that the process represented in »Figure 4.12 should be seen integrated in a larger framework as depicted in »Figure 4.11. However, one has to be aware that structural changes in a GMS take time. Hence, it is important as shown in »Figure 4.12 that processes are tailored to the current capacities and societal context conditions.

Capacity issues, professional and institutional competence (Step 8b)

Monitoring needs to be fit for purpose, taking account for resource constraints. Ecosystems respond to pressures and stressors in a variety of ways that can differ across water body types. Cost-effective monitoring, achieved through implementation of the cycle of planning, implementing, reporting and reviewing, use of external audit for quality assurance of monitoring and open access web-based platforms fundamentally depend on effective human capacity and institutional frameworks. This requires long term commitment of human and infrastructural resources.



Monitoring needs to be fit for purpose, taking account for resource constraints. Ecosystems respond to pressures and stressors in a variety of ways that can differ across water body types. Cost-effective monitoring, achieved through implementation of the cycle of planning, implementing, reporting and reviewing, use of external audit for quality assurance of monitoring and open access web-based platforms fundamentally depend on effective human capacity and institutional frameworks. This requires long term commitment of human and infrastructural resources.

Automated procedures for measuring hydrological and chemical components in the field can be highly effective, but require attention to maintenance. Unreliable measurements are useless. All cost related decisions need competent processes, and depends on balancing the usefulness of information to support management of ecosystems. This includes making judgments on the redundancy of measures and metrics.

The management cycle integral to water quality monitoring and reporting, with its well documented procedures for measurement, quality assurance and review (see »Section4.6.4), requires the necessary infrastructure to collect and process data, and effective management and professional capacity of human resources. Hence, all stages of the process from monitoring design to review of reported results rely on professional capacity and the enabling environment to develop skills and competencies. Inadequate human capacity is a frequent restriction to effective water management. To achieve and develop competent human resources requires appropriate recruitment and training procedures, and the professional environment for human capacity to act, reflect and adapt should be in place. It requires a long-term vision, that anticipates and plans for the required technical and relational skills and competencies within monitoring organisations.

There is a need for a wide range of managerial, financial and communication skills. This applies as much to development and implementation of WQGs as it does to other aspects of catchment management. Therefore, monitoring procedures are most effective when they are nested within well formulated management systems, with the commensurate skills and competencies of human capital employed at each stage, and with effective oversight of the entire process (Castells, 2008; Klijn et al., 2010; Newig et al., 2010). Adoption of professional management procedures such as project management certification (e.g. ISO 21500), quality management (ISO 9000) and Environmental Management (ISO 14001) supports this goal. Professionally managed institutions enable individuals to develop skills and values that provide clarity of their role in the overall cycle of monitoring and water management. The overall monitoring structure is greatly facilitated by descriptors of competence for individual professionals that define the knowledge and understanding needed to perform effectively (Cheetham and Chivers, 1996, 2005). Effective river basin management, or relevant sub-scales, requires social and relationship competencies as much as technical ones, an enabling environment of formal and informal institutions, informed political community willing to act, and ultimately the support of functioning civil society (Saleth and Dinar, 2005; Coria and Sterner, 2011; OECD, 2011). This, furthermore, requires bridging the gap between science and policymakers/ politicians, with dedicated actions designed to do that.

Professional competence

Human capacity needs relate to both technical and management skills. Technical skiils required for river basin management and monitoring include GIS, analysis of large data sets, modelling, water and biotic assessment and quality control, engineering, hydrology, and web-based platforms. Managing complex basins also requires high level of communication and human resources management skills, with negotiation a necessary addition to the repertoire when this includes involving multiple sectors, and even transboundary governance (Jarvis & Wolf, 2010). Effective monitoring benefits from a reflective process that facilitate social learning through multi-stakeholder dialogue (Pahl-Wostl et al., 2008; Dewulf et al., 2005). These processes can appear slow and inefficient (Partzsch & Ziegler, 2011), but generally necessary for sustainable water management (Allan, 2003; Warner et al., 2008). Building, and maintaining, these competencies requires careful planning, and diagnostics of current and future individual and institutional needs.

Competencies for river basin management, and inherently the water quality monitoring and standards that underpin that, are obtained and maintained through both formal and informal learning. A base of disciplinary skills provides a foundation of competence for an individual's input to the monitoring and mangement cycle. On-the-job experience and further targeted skills development builds the capacity to adapt and contribute to multidisciplinary awareness and problem solving. The formal learning of individuals involved in water management typically comprises initial qualifications across core disciplines outlined in the paragraph above, plus on-going relevant skills development, often referred to as “life-long learning”. Whereas this can be achieved in an ad-hoc manner, it can also be more formally designed based on institutional and individual needs; and developed through tiers of competencies that can be formally recognised with the career profiles of individuals. The tiers can cover basic technical skills to increasingly sophisticated inter-disciplinary and management skills. Skills recognition can be further formalised through certified professional training and accreditation of qualifications. The two components, competency and accreditation, respectively refer to demonstrable quality of work across a continuum from novice to expert, and formal and independent recognition that as institution or programme meets certain predefined quality standards.

Examples of certified professional competency includes attaining a standard of skills through accredited short or on-line courses, formal reporting on-the-job projects, workshop activities and training of others. Attaining an international network of professional competency for water quality monitoring and reporting relies, therefore, on:

- availing of appropriate education and training;
- the institutional framework that supports proficiency; and
- affirmation, perhaps through formal procedures, that confirms the necessary characteristics of individuals and organizations.

Institutional competence

Competent individuals are greatly restricted if they work within ineffective institutions. Duplication across different institutions has been a common inefficiency in catchment and water resoruces management. Effective water management requires functional institutions able to implement management and governance tasks. This can be greatly facilitated, and may fundamentally require, river basin or catchment authorities that can effectively oversee the activities of authorities responsible for different aspects of basin management. Commonly recognizable standards across organisations charged with water quality monitoring are enabling an mechanism for benchmarking, monitoring and reporting water quality. This, therefore, requires a set of institutional standards within which individuals operate, and which include effective human resources policies to fill gaps, and plan strategically and financially for current and future needs. The individual professional is, therefore, nested within an institution, which is nested within a larger and fluid arena of interacting organisations. Analysing institutional functionality in detail is outside the scope of these guidelines, but is a crucial factor in achieving cost-effective and reliable monitoring, reporting and ultimately acting on those results.

Explore regional cooperation and data/information sharing

Water management is a shared activity across different agencies and political boundaries. This can lead to establishing legal agreements within sovereign states of federal structure (e.g Federal Republic of Germany, US) and between states (e.g. EU, Rhine and Danube Basins etc). Where neighbouring jurisdictions have different traditions in use of methods for monitoring either a harmonisation of methods, or an intercalibration process (as in the EU) facilitates cooperative monitoring and data sharing. For many new transboundary basin authorities (e.g. agreements made for some of the large African lakes and river basins) agreeing in common methods to sampling provides very useful benefits.

Table 4.14: Competencies related to each of the management steps identified in »Figure 4.2. Levels (1-3) refer to (1) basic disciplinary to (2) advanced integrative skill sets or (3) responsible person(s), typically involving functional teams of people, with technical (T) or social (S) skills.

Phase	Step	Summary indicative activities	Respective common competencies and levels
Initiation	1	Vision, setting time-bounded objectives, coordination, bridging the gap between science and policymakers/ politicians	Management and integrated planning and communication, interdisciplinary and transciplinary knowledge and multi-sectoral awareness (T, S) Level 2 and 3
	2	Classification of water bodies	Biogeophysical, mapping; and satellite imagery (T) Level 1 and 2
	3	Collation of geo-spatial data sets, locations and ownership	Biogeophysical (T), Level 1
Identification, desk-top screening	4a	i) Putative indicators, pressures, stressors and risk assessment; ii) Stakeholder views;	i) Physical, chemical and biotic assessment techniques, spatial models (T) Level 2; ii) Communication and public participation (T, S), Levels 1 and 2;
	4b	Identifying high value and protected areas	Biogeography, water supply, protected policy and management (T,S), Level 2.
	4c	i) Present state of water bodies; ii) Application of indicators (T); iii) stakeholder engagement	i) Inventory relating state to spatial distribution (mapping) (T), Levels 1 and 2; ii) Ecological dose-response relationships; toxicology, sediment dynamics (T) Level 2; iii) Communication and public participation (T, S), Levels 1 and 2;
Assessment	5a	Production of Guidelines and field verification	Temporal-spatial sampling design and production of standard operating and assessment protocols (T), Level 2.
	5b	i) Establish guideline values; ii) stakeholder engagement	i) Applying dose-response relationships to geological characteristics of water bodies, statistical analysis for confidence (T), Level 2 and 3; ii) Communication, public participation and negotiation (T, S), Level 2;
	5c	Assign water body/subunit to ecosystem state	Assessment and aggregation of indicators judging ecosystem state (T), Levels 2 and 3
Monitoring	6	i) Sampling, ii) Synthesis data, identifying data gaps	i) Applying sampling and quality assurance protocols (T), Level 1 and 2; ii) Geospatial data analysis and (T), Level 2.
Evaluation, category assignment, reporting	7	i) Assessment of water quality at catchment scale within specified categories, ii) reporting, iii) linking to monitoring goals and, where necessary, refining monitoring regimes.	i) Assessing state of water bodies and uncertainty (T), Level 1 and 2; ii) and effective reporting and public dissemination (T, S), Level 2; Recommending changes or further quality assurance in monitoring (T), Level 2 and 3.

Policy Development	8a	Adaptive management and stakeholder engagement, Bridging the gap between science and policymakers/ politicians	Management and integrated planning and communication (T, S) Level 2 and 3;
	8b	Governance, legal framework and capacity	Engaging with government and citizens to evaluate the policy framework, river basin governance, and resource needs (T), Level 3.
	9	Securing funding, budgeting, cost benefit analysis	Economic , compliance to regulations, and stakeholder analysis (T, S), Levels 2 and 3.

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[60] http://ec.europa.eu/environment/water/water-framework/objectives/status_en.htm#_The_intercalibration_exercise

Cost benefit consideration and funding issues (Step 9)

N.B. The issues discussed in this step are mainly addressed in the 'Assess capacity' step, and the 'governance band' in the revised Framework (See Preface).]

Monitoring is only a part of the activities (and correspondingly that of expenses) within the realm of water resources management to safeguard and to rehabilitate freshwater ecosystems. However monitoring is a recurring expenditure, almost independent of the quality status of water bodies to be observed. These annual expenses are to be spent already prior to any well-designed and science-based freshwater ecosystem rehabilitation effort in order to assess the scope and potential foci of the tasks ahead. During the transitional period of restoration of impaired freshwater ecosystems monitoring has an important function to guide restoration measures and to document successes achieved. Once a freshwater ecosystem is restored monitoring is needed to ensure that the targeted quality standard is maintained. Securing long-term funding plays an important role in the context of monitoring and also for the guideline implementation in general.

While the total budget needed for freshwater ecosystem restoration is very difficult to estimate as location, size, severity and duration of deterioration and the attributes of the targeted (improved) state co-determine costs and schedule of the improvement efforts there are reasonable estimates for monitoring expenses.

For example, a small European country, which joined the EU about a decade ago and hence implements the EU WFD, spent slightly over 3 Million Euro in 2014 for water quality and freshwater ecosystem monitoring. In the same time a medium size developed Asian country monitored the integrity of aquatic ecosystems in streams and estuaries on an annual budget of 2.2 Million US$. Monitoring was carried out in 1000 sites twice a year assessing benthic diatom, invertebrate and fish communities. For estuary monitoring about 270,000 US$ was spent annually covering 30 sites. The same communities were observed twice a year.
Cost estimation of the Slovak Frame Monitoring Programme (2010-2015) covering surface water (quality and quantity), ground water (quality and quantity) and partly protected areas (e.g. sensitive areas, sources of water for human consumption) (»Table 4.15). Four public institutes participate in the Monitoring Programme that are managed by the Ministry of the Environment of the Slovak Republic. The costs include surveillance, operational and investigative monitoring according to the Water Framework Directive.

While the different context and monitoring obligations do not allow a comparison between these three examples, they illustrate well the order of magnitude of annual costs for high quality monitoring programmes in small and medium size countries.

Involvement of citizen science and school classes in monitoring programmes could extend the scope of monitoring and potentially contribute to reducing the costs of individual observations but it cannot replace professional services. Without a highly professional data evaluation, laboratory capacity, archiving and reporting and without professional guidance and support voluntary efforts cannot even be launched.

Monitoring also plays an important role in detecting when the condition of a water body is declining. Ideally, managers can discover what is causing the decline and take action to end, and even reverse the trend. This is very important because the cost of restoring a watershed can be very expensive. For example, the United States has spent on average $1 billion per year on stream restoration since 1990 (US EPA 2012). These numbers are expected to rise as communities work to mitigate environmental problems. In this way, a robust monitoring program is a very cost-effect strategy for assuring that aquatic ecosystems will continue to provide the multitude of services required to meet society's needs.

Table 4.15: Costs of the Slovak Frame Monitoring Programme (2010-2015).
Direct costs and investments (in Euros - rounded to 1000s).
Source: Compilation provided by the Slovakian Water Research Institute in 2015 based on information of the Ministry of the Environment of the Slovak Republic.

	2010	*2011*	*2012*	*2013*	*2014*	*2015*	*Total*
Direct costs for all institutions responsible for Monitoring Programme	*8 432 000*	*8 263 000*	*8 502 000*	*8 774 000*	*9 055 000*	*9 345 000*	*52 371 000*
Investments (e.g. analytical techniques, sampling devices, reconstruction of drillings for groundwater) for all institutions responsible for Monitoring Programme	*2 738 000*	*2 488 000*	*608 000*	*199 000*	*203 000*	*308 000*	*6 544 000*

5 Stepwise design and implementation of initiatives to improve aquatic ecosystem health: illustrative case examples

N.B. Many of these case examples are summarized in Volume 3, in which the steps have been updated to reflect the revised Framework (See Preface).

Context

The various examples presented in this chapter are intended to illustrate some of the steps taken in various projects and programmes that align with the sequence of four phases and nine steps proposed in the present Water Quality Guidelines (WQGs) for freshwater ecosystems (as described in »Chapter 4). All of the examples represent a retrospective view, in that they at least in part pre-date the guidelines. However, they each illustrate, to varying degrees, the extent to which on-ground practice has adopted or built upon a similar process or set of activities over time towards improved water quality and ecosystem health.

For each example study, the individual stages of the programme of work and activities undertaken are matched with the corresponding steps in the current guidelines (as indicated by the step numbers in parentheses, where appropriate; see »Table 5.1). Depending on the specific example, one or more of the steps recommended in »Chapter 4 may be absent or amalgamated together with other steps. In these cases »Table 5.2 indicates indirect reference. Several of the cases examined describe monitoring procedures, a central element of the guidelines described in this report. Notably, various aspects of the guideline steps against which the following studies are mapped, are as useful for water management in general as they are for water quality management and monitoring.

The examples described are all catchment based and their varying catchment contexts are described. Each case highlights different geographical setting, resource investments and time scales. In some cases the examples present massive, decades long restoration efforts. Realizing that the Framework is also applicable to these specific tasks of water quality and ecosystem management emphasizes its general nature. It also illustrates which steps were effective ones, which innovative, and what the consequences might be of missing any of them. The following examples also serve different purposes. Historical accounts (Examples I and VI), description of ongoing efforts (Examples II, III and V), summary of individual cases which may serve as intrdduction for further in depth review (Example IV) and highlighting the practical use of an existing data base (Example VII). In the first illustrative example an in-depth look at the development of an action plan to tackle eutrophication in Lake Balaton in Hungary is presented (»Section 5.2: Example I). A case from Latin America, the Upper Tietê River Cleanup Program, São Paulo State, Brazil (»Section 5.3: Example II) describes the stages of an ongoing rehabilitation programme to improve the water quality status of the Tietê River, a system for which a water quality guideline has been established.

The example from the Strickland River system, Papua New Guinea (»Section 5.4: Example III) focuses on the requirements for compliance by a mining company in the Asia-Pacific region. In it the development and implementation of the monitoring program and accompanying ecosystem health report card for the river system with its diversity of different waterbody types is described. The example study indicates the importance of setting clear objectives, and shows ways in which thresholds of concern for ecosystem health can be established.

The North America example (Study IV, »Section 5.5) comprises a composite, comparative review of 17 different case studies of the successful application of technical tools and approaches for developing strong biological assessment programs, in terms of the extent to which each of them demonstrates the approach outlined in the Framework. A useful summary mapping of which of the U.S. case studies best illustrate and align with the various steps of the Framework is also given.

Example V (»Section 5.6) outlines the current national process underway to draft WQGs for ecosystems in Indonesia for the first time, with particular emphasis on the typology of local lakes. Lake Maninjau is selected for illustrative purposes showing recently emerging stressors and the potential steps to be undertaken to safeguard the lake ecosystem. Example VI (»Section 5.7) summaries the efforts made in the Republic of South Africa in the last two decades. The example reviews the development of the governance and legal framework, institutional and methodological development and includes an illustrative example (Olifants River) of the implementation so far. Example VII (»Section 5.8) illustrates how to develop a national diagnostic assessment of ecosystem health based on available archived water quality data in Mexico.

Ideally, this chapter of the guidelines can expand over time, as additional case study examples come to light that showcase the diversity of approaches that have been adopted worldwide.

Table 5.1: Overview of how the individual implementation stages of example studies match the 4 phases 9 steps of the Framework.
× = adressed
◯ = indirect references as the example is not a guideline development

Phases	Steps	I Lake Balaton, Hungary	II Upper Tietê, Brazil	III Stickland River, Papua New Guinea	IV 17 federal states, US	V Lake Maninjau, Indonesia	VI Olifants River, South Africa	VII National diagnosis of river basins, Mexico
Initiation	1	×	×	×	× No.14	×	×	◯
	2	×	◯	×	× No.02	×	×	×
	3	×	◯	×	× No.03	◯	×	×
Identification	4	×	◯	×	× No.04	×	×	×
Assessment	5	×	◯	×	× No.01	×	×	×
	6	◯	◯	×	× No.17	◯	×	◯
	7	◯	×	×	× No.07	◯	×	◯
Policy development	8	◯	×	×	× No.13	◯	×	◯
	9	◯	×	×	× No.08	◯	×	◯
Comments					see detail in Figure 5-16			

Example I: Lake Balaton, Hungary: Eutrophication management policy

The present section summarizes experiences of the preparation (1978-1982) and implementation (1983-2010) of the Balaton eutrophication action plan. It illustrates the extent to which alignment could be found between the Framework and those steps undertaken during the restoration of Lake Balaton. The Eutrophication Management Policy of the Balaton is a well-known case, targeting a lake which is not only the prime recreational area of Hungary, but also a Ramsar site.

Among other land uses and polluters, the wine growing and other agricultural areas in the Balaton region received in the 1960s and 1970s large amounts of fertilizer and pesticides inputs leading to a serious deterioration of the lake's water quality. Furthemore, unbalanced water infrastructure development (focusing on drinking water supply, but overlooking sewerage), large scale animal farms and inadequately treated wastewaters contributed significantly to the pollution problem. The situation is described as in the early 1980s together with its knowledge base, however a methodology of today (in fact for the discussion of technical issues the procedure described in »Chapter 4 is followed). Obviously, the objectives were not quite the same for the two cases: policy development for a specific water body and its catchment, and WQGs, respectively. In spite of this difference, the two procedures have many common elements and thus, it will be possible to demonstrate the versatile role of guidelines like the Framework in policy making.

Lake Balaton is a large (about 600 km²) and shallow (average depth is a little above 3 m), turbid, phosphorus-limited lake of high carbonate content with high alkalinity and high pH (»Figure 5.1). It has a narrow and elongated shape and approximately a W-E orientation. Due to strong wind action, the water is never standing still (the prevailing wind is NW-N). The largest inflow is the Zala River at the Western end, while there is a single outflow at the East where a sluice is regulating the water level (»Figure 5.1). Average water residence (filling up) time is about two years. In the present context (nutrient loads and water balance of the lake) groundwater does not seem to play a significant role. The Northern side is hilly, while the Southern is rather flat. The catchment area, located entirely in Hungary, is about ten times larger than the lake. The first signs of man-made eutrophication were recognized at the 1940s by sporadic observations of scientists of the Balaton Limnological Research Institute (Entz and Sebestyén 1942, Entz and Sebestyén 1946) (»Figure 5.2). Later on, the change in the ecosystem was indicated by invasion of fibrous green algae, by the mass proliferation of Ceratium hirundinella (a freshwater dinoflagellate) and by regular observations of primary production, biomass and Chl-a data. The public only became aware of the danger threatening the lake later after two major fish kill cases in 1965 and 1975. The water had also assumed a greenish hue, and in some places decaying debris formed repulsive blankets. In August 1982 the massive proliferation of the sub-tropical N-fixing blue-green algae Cylinderspernopsis raciborskii had a striking impact (Herodek 1986). Peak Chl-a value was close to 200 mg/m³.

Figure 5.1: Lake Balaton and its (entirely domestic) catchment and larger settlements. Zala River: main tributary; Sió: outlet; red line: border of Hungary.
Source: Somlyódy and van Straten (1986).

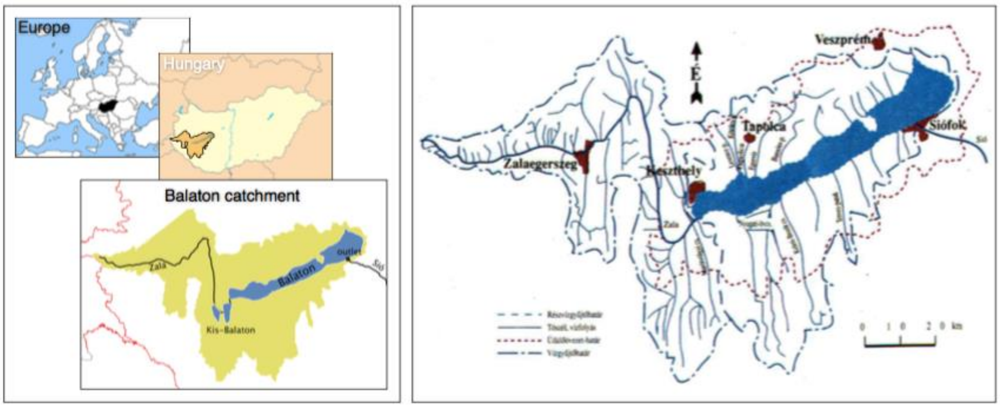
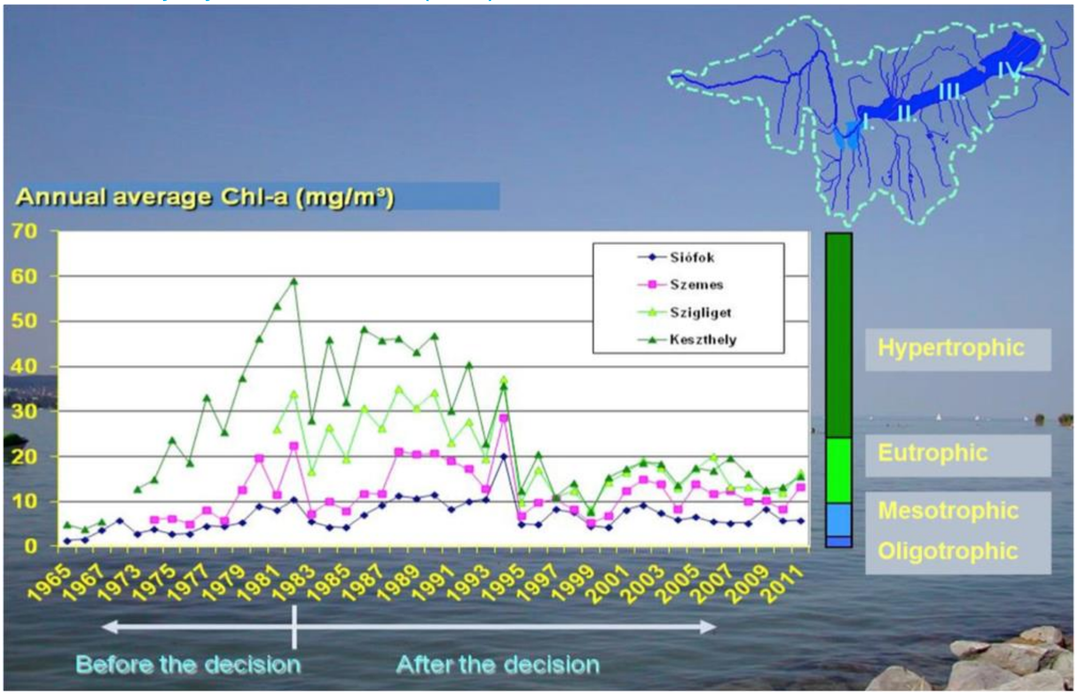


Figure 5.2: Trophic state of four basins (I-Keszthely, II-Szigliget, III-Szemes, IV- Siófok) of Lake Balaton.
Four basins depicted in the upper right corner; data for 1965 – 2011.
Source: Somlyódy and van Straten (1986).



Practically the entire lake became hypertrophic (excessively enriched with nutrients). Subsequently, the restoration of the lake became a national concern. A policy-making procedure was initiated by the Government (»Box 5.1) and a strategy for improving water quality was prepared and approved in January 1983 (Láng 1986).

Following the major steps of the FrameworkIWQGES procedure, next it is discussed what happened with Lake Balaton and its catchment at the early 1980s.

The ultimate goal of the Lake Balaton restoration strategy was to attain and preserve water quality levels that prevailed in the early 1960s. It was recognised that this would take up to several decades to attain. Thus, the process of implementation was subdivided into several stages. First, the targets had to be specified. They should express not only the goals of the water quality control as a stepwise procedure, but they should also be easily understood by "outsiders". Eventually, three levels of water quality (trophic state) target conditions (A, B and C) were defined, as follows:

Level A. Conservation of the water quality of the late 1970s and early 1980s, i.e. prevention of further deterioration. This, however, would not exclude the possibility of exceptionally high algae production levels under adverse conditions, such as hot, rainy summers.

Level B. A period of gradual improvement, in which the appearance of high algae production levels can he excluded with high probability.

Level C. Restoration of water quality of the early 1960s.

Although, trophic states can be unambiguously assigned to levels A to C (e.g. for Basin I hyper-eutrophic, eutrophic and at least mesotrophic or prescribing Total Phosphoroous (TP), Biologically Available Phosphorus (BAP) and Chlorophyll-a (Chl-a) values for the four lake basins) the use of technical jargon was avoided. The definition of A, B, and C levels have proven successful and effective.

The schedule for the attainment of levels were specified, in the frame of adaptive management, as follows: A, B, and C - 1990, 1995-2000, and 2005-2010, respectively which took into account the delayed response due to the sediment. Namely, the internal P release from the sediment was roughly equal to the external load. For this reason the pre-condition of a significant improvement in the trophic state is the renewal of the sediment which was estimated to about a decade. The attainment of water quality level A meant that a wide spectrum of mainly technical measures had to be implemented from P precipitation, sewage diversion, pre-reservoirs, erosion control, encouragement of using "soft" detergents and changing farming practices, temporary stop of any further expansion of tourist accommodation along the lake. These proposals formed an "integrated program package" of a tight deadline of 1987. Namely the fear was that the progress of eutrophication cannot realistically be expected to be arrested otherwise, i.e. targets set will not be achieved. The detailed methods for attaining water quality levels B and C have not been developed to the same level of detail as those for level A; this would have been unrealistic since they would largely depend on the effectiveness of measures of level A to be monitored and evaluated in an adaptive management framework.

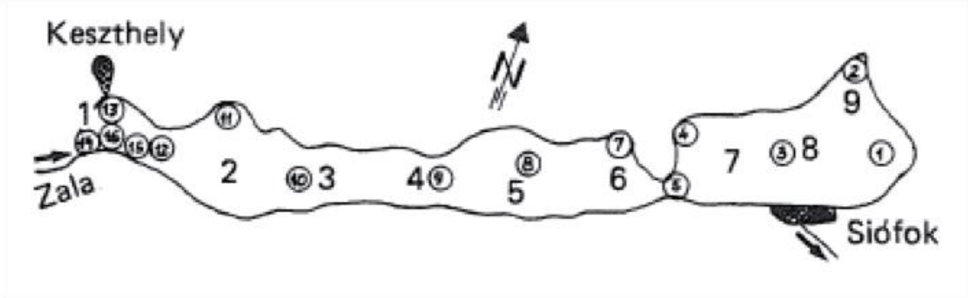
Box 3 'No net loss' policies for wetland extent.

Setting objectives (Step 1)

- The long-term vision was to achieve a trophic state similar to that of the early 1960s (stakeholders still remembered how the lake's water looked like) and improve the the microbiological quality in the near shore area.
- Future trends of tourism, agriculture, industry, urbanization, water infrastructure development and water management were analyzed. Desired changes were identified in the frame of integrated program planning (it is noted that trends have drastically been changed due to the political change in the country in 1989 - 1990).
- Short-term and longer-term goals were formulated for the lake and its segments in terms of BAP and TP loads influencing primarily short term- and long term behaviours, Chl-a concentrations (annual- and summer period average values, as well as summer peak values were tested as indicators, see below). Disinfection[61] at wastewater treatment plants (and 1st quality class

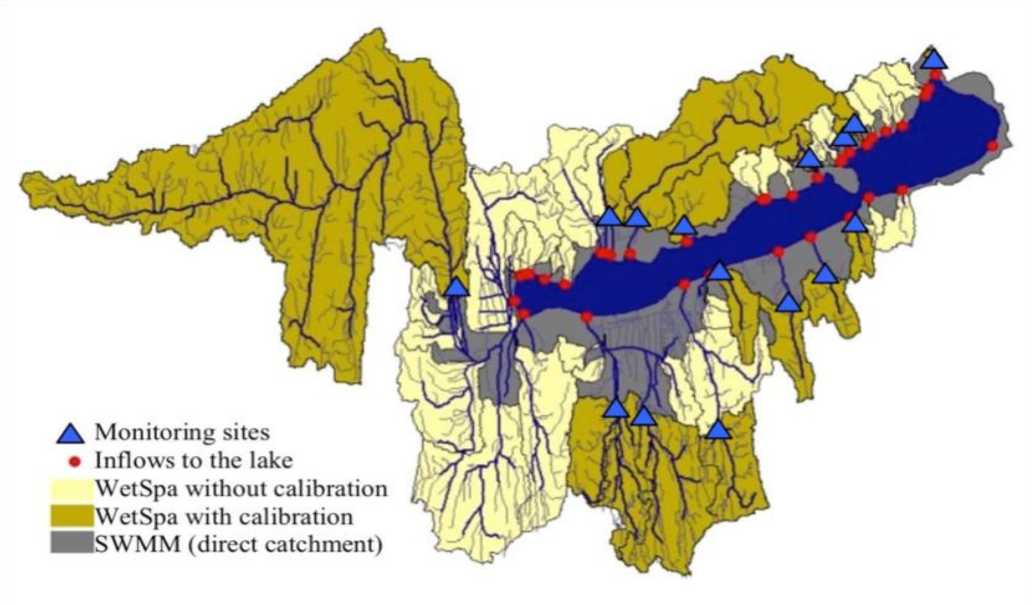
[61] As Lake Balaton is a recreational area treated sewage water was disinfected before discharging it into the lake.

bathing water) was added as a criterion. Algae biomass and structure associated were used for cross-checking targets. They are sensitive and informative parameters, the monitoring of which is however too time consuming (and expensive). It is noted here that a pre-impoundment at the Zala River mouth at the Western end of the lake (»Figure 5.1) was designed as an element of the P control plan leading to hydro-morphological and habitat changes. It can be considered as the partial restoration of the Kis-Balaton and Lower Zala Valley wetland (Szilágyi et al. 1989) which belonged to the lake prior to its drainage at the first quarter of the 20th century.



a)

Figure 5.3: Monitoring networks of Lake Balaton.
a) Lake water quality, b) Watershed, inflows, catchment with/without measured loads.
Source: Istvánovics et al. (2007).



b)

- Monitoring objectives included to detect trophic states - particularly during the summer recreation period, monthly nutrient loads of tributaries and loads of diffuse sources of agricultural and urban runoff origins, and obviously point sources (»Figure 5.3a). Also, the integration of two lake water quality (physical and chemical) monitoring networks was an issue together with determination of stratified sampling (from monthly sampling during the off-season to twice a week during the summer period). One of them was run by the water research institute VITUKI included 9-centerline locations, while the other, operated by the Transdanubian District Water Authority, also had transversally distributed points (altogether 16, »Figure 5.3b).

- A special focus was given to the largest tributary, the Zala River where since 1977 daily observations were ongoing. This unique data set was the guarantee to derive a credible load estimates for the entire lake. Also, it served as a basis for modelling rainfall-runoff, erosion, diffuse loads etc. and catchment planning

(Somlyódy and van Straten 1986).

- Additionally, a number of research programs and associated modelling efforts were done to improve the understanding (primary production, algae nutrient limitation, sorption, the role of wind and suspended sediment on light penetration and algal biomass, sediment, internal phosphorous load etc.) and for estimating the chemical and ecological status (as it is said today) of Lake Balaton.

- TP, BAP and Chl-a were logical choices to serve as indicators. There were others, frequently used ones which have been tested, such as DO, transparency, particulate P, etc. However, their application as an indicator turned out to be less useful since their level was primarily determined by winds induced mixing and re-suspension (and PO4-P remained relatively constant).

- No general objectives were set for reporting and communication till 1982, when three groups of stakeholders were identified and involved: scientists, the public (mostly from the direct vicinity of the lake) and decision makers/politicians including the top level.
- Objectives were defined for quantifiable measures to be taken (not forming a part of developing WQ guidelines). Examples include upgrading level of P removal at waste water treatment plants (WWTPs), capacity extensions, diversion of wastewaters from the catchment, sediment dredging, construction stop in the recreational area etc. For not quantifiable measures, desired trends were indicated (change to other farming methods, control of agricultural inputs and erosion in large scale farming (agricultural co-operatives), use of P free detergents etc.).

- Water quality, nutrient load goals and measures were scheduled for the 1983-2010 period in three stages, keeping in mind the lag effect of the internal load associated with sediment P release (see »Box 5.1).

Classification of the lake's ecosystem (Step 2)

- From the viewpoint of limnology, hydrology and data base organization the lake can be subdivided into four consecutive basins/segments (»Figure 5.2). The smallest Western basin (I) is pressured by the largest catchment area while for the largest Eastern basin (IV) the opposite is true. This feature suggests a strong longitudinal W-E gradient in water quality.

- The climate is semi-arid. With high annual temperature fluctuation, summer water temperature can reach 30 °C. On average, the lake is frozen over for two months in winter. Due to strong wind action, a complex, unsteady flow pattern is generated with high suspended sediment concentration in the water. Thermal stratification almost never occurs.

- The concept of typology of the water body was unknown in the early 1980s. The distinction of four lake-basins would correspond today to four water bodies of differing ecosystem states and seemingly also differing reference conditions (RCs). However, it should be kept in mind that the distortion of the ecosystem was a response to the excess amount of nutrient loads for a long period of time. During the 1940s, the ecosystem of the lake was uniform and healthy, and water quality was excellent (see »Figure 5.3). This 1940s status is exactly our future the goal that was set in the 1980s. Thus, the four basins of the lake (water bodies) belong to a single type and a single RC. Approaching one common RC for the entire lake can serve as a future goal. Another option would be to identify the lake as a single water body, which however would cause methodological difficulties to handle properly the role of mixing.

Setting the basin-scale context (Step 3)

A sequence of maps were used which included counties, district water authorities, various levels of sub-catchments and tributaries, recreational areas subject to fast development and high pressures and consequently to a number of stressors during peak season of tourism, locations of monitoring networks, land uses, soil and sediment composition, agricultural cultivation, fertiliser application, location of large scale animal farms and settlements etc., which allowed estimation of nutrient sources at the site of generation and their portion reaching one of the basins (see »Figure 5.3 as an illustration). Various monitoring-, pilot scale-, event based programs and modelling efforts served the gradual refinement of the estimates. Detailed maps were also available for the Kis-Balaton area being a Ramsar site (connected to the Lake Balaton Ramsar site and later belonging also to the NATURA 2000 network).

Desktop screening (Step 4)

By the late 1970s not only physical and chemical water quality data were available, but also biological observations including phytoplankton, zooplankton, fish, etc. However, their systematic analyzes remained missing. The first data base paved the way from simple time series analyses to sophisticated multi-dimensional water quality simulation and management optimization modelling. As a first step multiannual and lake wide averages of various water quality parameters were derived in the frame of the comprehensive lake – watershed approach based on the concept of decomposition and aggregation (Somlyódy, 1982). Simple statistical analyzes and empirical modelling performed under the assumption of complete mixing did not lead to any meaningful conclusions, as the underlying mixing hypothesis is not fulfilled in case of a hydromorphology like that of Lake Balaton.

Screening of pressures and stressors (Step 4a)

Second, the scope was broadened using the well-known methodological framework: Drivers, Pressures, Stressors, State, Impacts and Response (DPSSIR). Also, the lake was subdivided into four consecutive lake-basins or segments (as completely mixed reactors in series) well-suited for mass balance calculations and Volleweider-type of simple phosphorous and nitrogen modelling (OECD 1982) hoping to be able to account for mixing more realistically than before.

Drivers, pressures, stressors, processes determining pressure – impacts (P – I) pathways and potential impacts are summarized in the first four columns of »Table 5.2 (see also »Section 2.4). As can be seen impacts related to nutrients, eutrophication and ecosystems are dominating. The last column of the table represents changes and trends. All the DPSSIR elements and indicators are expressed for the four lake-basins separately, since there is a large difference among them in volumetric phosphorous loads, residence time as well as the ecosystem structures.

Future threat and risks include consequences of lack of knowledge, of not performing desired assessments, the poor characterization of the present state, settings of goals too optimistically or pessimistically, the further increase of phosphorous and nitrogen loads and concentrations, which would have had a detrimental impact on the already hypertrophic state, internal load, phytoplankton composition, the dominance of nitrogen fixing blue-green algae being able to form toxins. These would have hampered rehabilitation or much longer time period would have been needed than under the current conditions. Also, there could be surprising synergetic interactions etc.

Table 5.2: Drivers, pressures, stressors and impacts of Lake Balaton at the early 1980s.
The last column indicates changes between 1960 and 1980 or pressures in the year given, unless noted differently.
Notes: * also stressor; ** State in the given year; 1985 is indicated as the time of broadly recognizing climate change as an issue. The first study was made at the early 2000s after a three-year long drought period when the question of transferring water from another catchment was addressed.
Source: Somlyódy and Honti (2005).

Driver	Pressures and their manifestations	Stressors and their manifestations	Impacts	Change/ trends

1. Agriculture	• *Terrestrial biomass production (food, timber, energy crops, animal husbandry and fish ponds etc)** • *Biomass extraction (e.g. fishery and aquaculture)* • *Water withdrawal/ discharge(agricultural use)*	• *Biological pollution (invasive species, pathogens, etc.)* • *Chemical pollution* • *Modification of aquatic habitat (Habitat degradation & loss)*	• *Nutrient levels, eutrophication and ecosystem changes for the four basins* • *Toxicity* • *Alien and invasive species. Impact on native fish and nutrient cycling* • *Ecosystem changes*	• *6-7 x increase in fertilizer application* • *1960- 2000* • *1920s (drainage of the Small-Balaton wetland)*
2. Industry	• *Water withdrawal/ discharge (Industries including extracting industries)** • *Climate variability and change, atmospheric depositions**	• *Chemical pollution*	• *Nutrient levels, eutrophication and ecosystem changes for the four basins* • *Pollution*	• *1980***
3. Mining	• *Water withdrawal/ discharge (Industries including extracting industries)**	• *Flow alteration* • *Overexploitation*	• *Dilution, water balance and level*	• *1960- 1980*
4. Energy, transportation	• *Climate variability and change, atmospheric depositions**	• *Modification of aquatic habitat* • *Biological pollution (invasive pathogens, etc.)* • *Chemical pollution* • *Thermal pollution*	• *Most of the impacts listed for the other drivers*	• *1985-*
5. Urbaniation	• *Transport, infrastructure and traffic** • *Water withdrawal (domestic use)* • *Human settlements (especially In the proximity of water bodies)*	• *Biological pollution (invasive pathogens, etc.)* • *Chemical Pollution*	• *Nutrient levels, eutrophication and ecosystem changes for the four basins*	• *Water Supply:5 x increase*
6. Tourism	• *Human settlements (especially in the proximity of water bodies)** • *Recreation* • *Navigation (including rafting/ floating)* • *Hazard security (flood protection etc.)*	• *Water Infrastructure* • *Modification of aquatic habitat* • *Chemical pollution*	• *Nutrient levels, eutrophication & ecosystem changes for the four basins* • *Ecosystem changes* • *Toxicity* • *Alien & invasive species. Impact on native fish and nutrient cycling* • *pollution*	• *About 150km length of constructed shoreline* • *14 x increase, 8 million visitor days in 1980*

Pressures originate from the consequences of activities undertaken to achieve human aspirations and can be characterized by various indicators or indices as done in the frame of the “watershed development approach” for Lake Balaton and its catchment at the early 1980s which used quite a number indices (erosion potential, fertilizer use, population density, density of near-natural water courses etc., Dávid and Telegdi, 1986). Since indices demonstrated similar changes with time, the approach remained non-conclusive. Stemming from »Table 5.2, the pressures and impacts can be well characterized as indicated in »Table 5.3. By far the most important impact is eutrophication. Its indicators are (monthly or) annual average TP and BAP loads, volumetric phosphorous loads (being also pressure indicators) and annual (and summer) average Chl-a. The reliable estimation of these quantities is a key element of policy making. Nutrient cycle models were used in a sensitivity analysis fashion to estimate for a given sampling frequency the length of nutrient averaging periods for tributaries such that errors and uncertainties remain in an acceptable range. The conclusion was that sampling once a month was quite feasible.

Table 5.2 and Table 5.3 identify quite a number of pressures/stressors. Thus, the question should be addressed on how was it determined that eutrophication was the most important impact? The answer may not be straightforward: even at the early seventies there were serious debates of not rarely subjective elements on the nature of the problem. For instance, several scientists and policy makers argued that organic material pollution was the main issue, although DO conditions in the lake remained permanently excellent. As the assessment considered, there were two important steps.

Source evaluation: emission – transmission – immission estimates were performed for the watershed and compared to water quality components of basins. Temporal changes were also accounted for. Between 1960 and 1975 tourism, fertilizer application and large scale animal farming increased by a factor of 15-50, drinking water supply grew so that by 1975 the demand could be completely satisfied while sewerage and Waste Water Treatment (WWT) development did not keep up. Among various pollutants a “harmony” or correlation among watershed and load data was achieved only for nutrients, demonstrating the primary role of impacts agriculture, urbanization and tourism (»Table 5.3). Furthermore, phytoplankton structural changes made it evident that the issue was eutrophication.

Order of magnitude estimates were made and evaluated for various pressures. Industry in the East was found to contaminate the atmosphere which contributed via atmospheric deposition contributed to about 10 % of phosphorous loads of the lake. Similarly, mining primarily had a local impact on some of the tributary flows in the north and less than 10 cm increase on lake’s water level (for comparison, the natural fluctuation is above 1.5 m).

As other pressures and stressors are considered, heavy metals are atypical in the region. Pesticides contributed to two major fish kills which led to a shock of the public. The reaction was serious pesticide and erosion control, and encouragement of the changeover of agricultural practices.

Estimate present state of the inland water ecosystem (Step 4c)
From pressure and stressors screening it became evident that the main problem of Lake Balaton is formed by excess amount of nutrients. As noted, unusual phytoplankton structural changes were already observed during the early 1940s, but causes were not known. Routine physical and chemical data were available from the late sixties. Quite a number of additional data bases (e.g. soil, vegetation, land uses, »Figure 5.4) were there, too, which have never been used in an integrated fashion to analyze the complex, interdisciplinary problem of the lake. The procedure adopted was as follows:

A simple data base was created which included water quality, main parameters of hydrology, meteorology, climatology, etc. as a function of years and days furthermore most important information of various maps. Various estimates were derived systematically and key questions were addressed such as: is it really eutrophication? If yes, what is the limiting factor? Any cause-effect relationships? Are there pronounced spatial and temporal alterations? If yes, why? What to control and how? From the data available, average TP and Chl-a concentrations were calculated for the period 1976-1978 and plotted along the longitudinal axes of the lake – practically along the sampling points. A pronounced W – E gradient was found which showed a 3 to 10 times increase, depending on water quality parameter (see »Figure 5.2 for basin averages).

As land uses were considered, the W sub-watershed was dominated by intensive agriculture while the other end was pressured by tourism. From this pattern it was assumed that the load distribution was more or less uniform longitudinally and thus the geometry of the lake largely defines the decrease of volumetric phosphorous loads of the four basins from West to East, clearly supporting the longitudinal change demonstrated by water quality data.

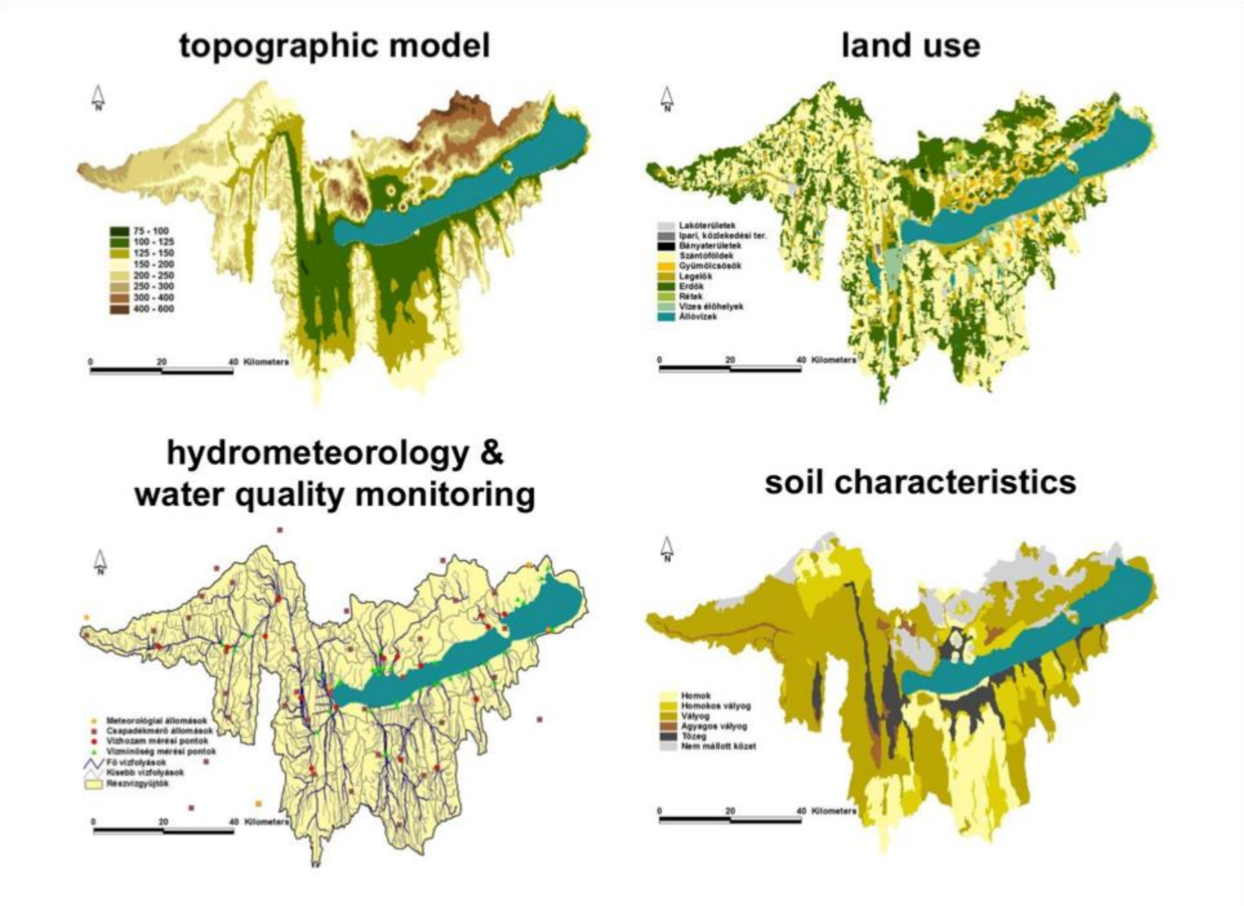
Pressure screening led to a number of conclusions:

iv. due to intensive agricultural activities in the basin the main impact is nutrient enrichment and eutrophication ,

v. there should exist a relation between Chl-a and external load (being one of the pressures), though it may not be well known,

iv. due to the highest volumetric phosphorous load, Basin 1 is in the most critical hypertrophic state, particularly since there seems to exist a W to E proliferation in the process of eutrophication (which was analyzed later on by a coupled 2D hydrodynamic - transport - water quality modelling, Shanahan and Harleman 1986).

Figure 5.4: Input Geographic Information System (GIS) data to a nutrient catchment model: topography, land use, monitoring networks and soil characteristics (as they are available and used today).
Source: Istvánovics et al. (2007).



Note, that until now very simple tools were used, only, to estimate basic quantities.

The present state of the catchment was also characterized as an element of integrated assessment. Subsequent to subdivision into sub-catchments, direct vicinity of the lake, recreational area etc., coherent nutrient balances were set by determining land uses, emission and transmission estimates (Somlyódy and Jolánkai 1986). Having the experiences at this stage, monitoring was changed in several steps. This included adjustments in frequency, sampling sites, estimation of (unit P) loads, event based measurements, sediment fractionation (Lijklema et al., 1986) and many others. This step included the development of simple and complex, dynamic phosphorous and nitrogen models, their usage to estimate the “phosphorous load reduction - algae biomass” short-term response function, the analyses of stochastic impacts (e.g. tributary loads, meteorological factors) and uncertainties on Chl-a, as a function of natural and man-made factors etc. (see »Section 5.2.6). In short, parallel to the policy issues there was an interlinked, adaptive monitoring and research program going on.

Sediment is an important factor in the lake's nutrient budget. It was confirmed during the early 1980s, via nutrient cycling models (Somlyódy and van Straten 1986). It was found that the internal load is about the same as the external one. Thus, on the short run an ideal, complete reduction of phosphorous loads would lead to 50 % biomass decrease, only. The recognition was new and timely. Experiments justified the results ten years later. The exhaustion of the internal load depends to what extent suspended solids diluted in P enters the sediment (sediment renewal). The typical time scale was found to be about 5 to 10 years. This leads to a "delayed response", meaning a lag in the lake' response to external load reduction. Any major improvement depends on the "renewal" of the sediment. In turn, any postponement of measures would probably result in an increase in the lag.

Establish water quality guideline values for indicators of concern (Step 5)

»Chapter 4 of this volume proposes to follow a two-tired approach. Here, the purpose is comparison and illustration, and thus a simplified Tier-1 approach will be followed.

Lake Balaton in Hungary is in a favourable position from the viewpoint of pressure and stressor impact indicator selection. Regular physical and chemical monitoring was launched in 1968. The book on biological classification of surface waters was published in 1974 (Felföldi 1974). Thus, experiences were there. This is particularly true for the lake: scientists of the Balaton Limnological Institute were studying the lake for nearly 100 years. Stemming from abiotic conditions and ecosystem structure, not more than two or three indicators are adequately characterizing the problem (though functional ecological indicators might be missing): TP (and BAP) load and Chl-a (expressed e.g. as % reduction/improvement ratio of the worst year). In terms of guidelines the widely used 4 (5)-Class Organisation for Economic Co-operation and Development (OECD) system (the lowest class is rarely used), a similar Hungarian system, and other schemes from the literature were also available (reference is made to the associated outcome of United Nations Educational, Scientific and Cultural Organization's (UNESCO's) Man and the Biosphere (MAB) program, Ryding and Rast 1989). One of the reasons of the broad offer of classification schemes is that although the expression "trophic" stems from the pioneers of limnology, still the interpretation is diverse and quantification is difficult. »Table 5.4 provides more detailed information on classification schemes.

Table 5.4: The OECD classification system, the proposedFramework scheme and individual basins of Lake Balaton in 1982
* Year when Basin 1 reached the trophic level indicated
Note: The table defines from left to right five classes as ranges of mean and max Chl-a, years when given classes were reached, the proposed Framework categories, the 1982 state, reference conditions and targets per basins.

Chl-*a* / Category	(Chl-*a*) (mean; mg/m³)	(Chl-*a*) (max; mg/m³)	Balaton Basin 1*	~~Framework~~ ~~IWQGES~~ Scheme	Balaton basins in 1982	Reference Conditions of basins	Present targets
Ultra-oligotrophic	< 1.6	< 2,5	~ 1940	**Cat. 1**			
Oligotrophic	< 2.5	8	~ 1965	**Cat 1**		**I, II, III, IV**	**II, III, IV**
Mesotrophic	2.5-8	8-25	~ 1972	**Cat 2, 3**			**I**
Eutrophic	8-25	25-75	~ 1974	**Cat. 3**	**IV**		
Hypertrophic	> 25	> 75	~ 1977	**Cat. 4**	**I, II, III**		

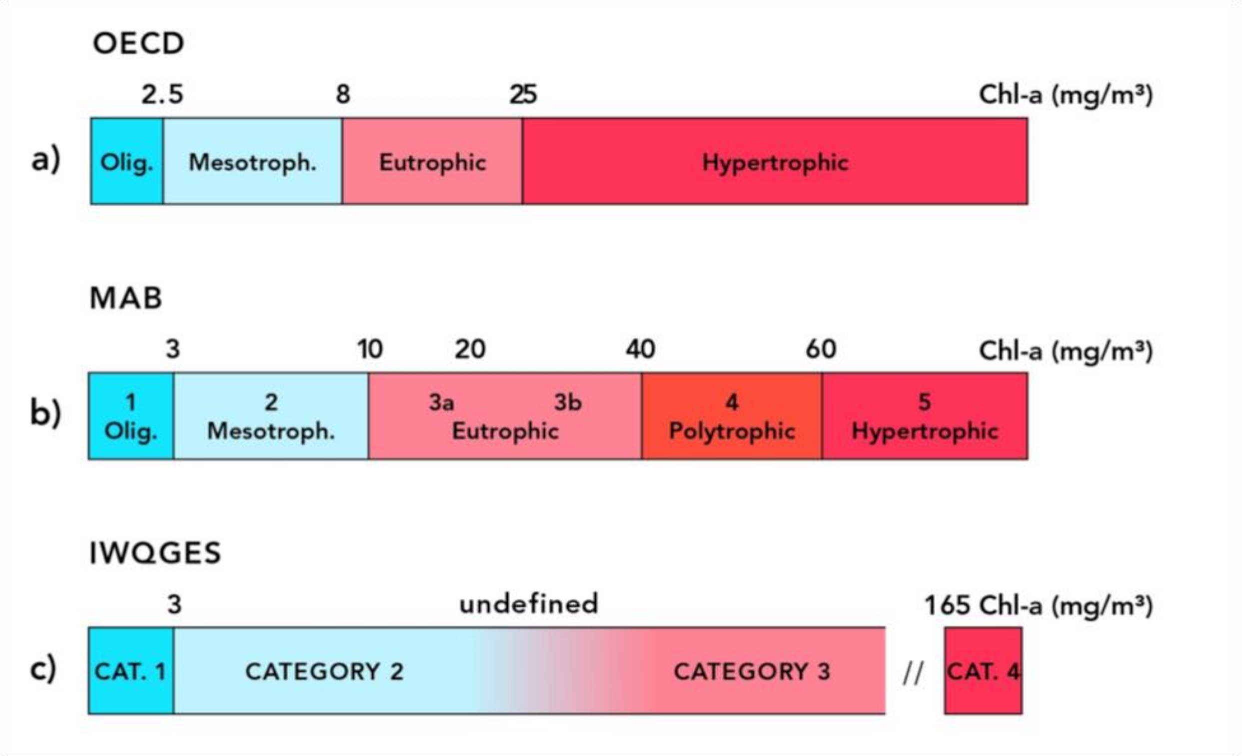
- The OECD trophic categories and threshold values are presented in »Table 5.4 and »Figure 5.5). From the gathered data and the information displayed in »Table 5.4 it follows that by 1982 - except Basin IV being eutrophic - the lake passed a threshold to a hypertrophic category. Using the 'Framework' notation one may say that Basins I-III of the lake were highly disturbed, hypertrophic, while Basin IV was perhaps moderately disturbed, eutrophic. All in all, these imply that there was a failure to achieve the tourism related water quality threshold values expressing a serious concern and thus there was a need for an urgent action.

- In addition to the fixed threshold system, OECD suggested also an open boundary system indicating the impact of uncertainties and subjective judgement: limnologists were asked to rank lakes from the viewpoint of trophic state and to consider threshold values as statistical distributions. This offers a gradual transition from one class to the other and probabilities of belonging to the given classes.

- »Table 5.4 also shows the estimated year when Basin 1 of the lake entered the class indicated and gives an impression on the temporal and spatial progression of eutrophication, which was identified as the overwhelming impact.

As a test, »Table 5.4 compares the classification approach proposed in the Framework (marked as “IWQGES” in »Figure 5.5) to the OECD and MAB schemes. A number of interesting conclusions can be drawn. First, it is obvious that indicators are not the same (annual average Chl-a and April – October averages, respectively), but they are in harmony with each other. Second, the number of classes is also differing. Third, the “high ecosystem integrity” class of ‘IWQGES’ approximately corresponds to the mesotrophic class (or vice versa). Fourth, the “eutrophic” class seems to be too broad (probably this is why the MAB scheme subdivides it). Fifth, the upper bounds of the mesotrophic and eutrophic classes seem to be the most significant ones. This would specify a simple three-class system. Sixth, the proposed four category system (‘IWQGES’) seems to be a suitable and flexible approach.

Figure 5.5: Comparison of classification systems.
a) OECD trophic state, annual average Chl-a; b) MAB trophic state, April – September average; c) IWQGES. Olig. = Oligotrophic; Cat. 1 and Cat. 4 = Water Quality Categories 1 and 4 of IWQGES. In c) proposed IWQGES benchmarkvalues for lakes and wetlands are displayed (see »Annex 2, »Table A.6); please refer to the same table for values for rivers. Source: Somlyódy (2015a). N.B. “IWQGES” refers to International Water Quality Guidelines for Ecosystems, as was adopted in earlier drafts of this report.



- In light of the above, it is up to the user on how to select guideline values, whether he or she accepts e.g. the OECD or the MAB scheme or following local conditions a modified version is prepared.

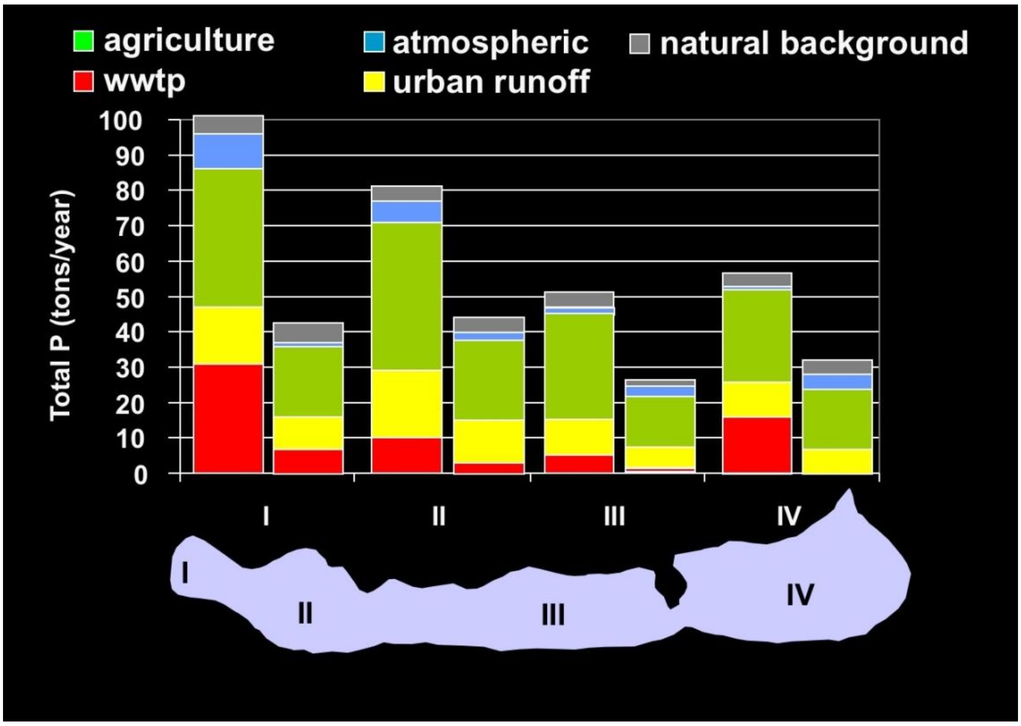
- RCs can be set by assuming – correctly or incorrectly - that the future may be similar to the past. Till the 1960s Lake Balaton demonstrated quite a uniform trophic state of a few mg/m³ Chl-a. But, is it really a feasible goal as well? The answer is probably "no" since phosphorous load levels may not be controlled at an extent as it might be set by the guidelines. The Lake's TP load in the early 1980s was about 300 t/y which was reduced until today by about 50 % (»Figure 5.6).

- If components of TP loads are considered (see »Table 5.2 for the similar structure), before the interventions sewage discharges, urban runoff, agricultural diffuse loads and atmospheric dry/wet deposition were the main contributors (for Basin I about 30%, 15%, 40% and 8%) on top of the geochemical background What can be seen is that more than 10% was not-controllable: in an ideal case this is the minimal load scenario. By calculating the volumetric TP load and speculating on the history of eutrophication this alone would result in for Basin I a state of mesotrophic-oligotrophic. A more realistic, BAC scenario would be to assume that sewage load practically disappears, and urban and agricultural diffuse contamination is halved in comparison to the "before" situation (»Figure 5.6). This would require further load cuts of the favourable "after" case (»Figure 5.6) characterized by four dry years in a row. It would gradually lead to an eutrophic-mesotrophic state in Basin I, mesotrophic-oligotrophic in Basin 2 and oligotrophic at the rest of the lake (in fact, this process is observed now – having many rainy years during the past decade – and it can also be observed how the four basins are approaching each other in terms of trophic state and phytoplankton structure). This applies on the short run while on the long run further improvements are anticipated due to the continuous renewal of the sediment and formation of $CaCO_3$, unless there is a phosphorous load increase.

- The Framework could be applied more or less as outlined above: water body identification, types and RCs (the "dream" or vision), future goals (phased) and their rationale, present state (water body + basin), screening from the quick, simple to complex ones according to needs, and setting guideline values for indicators of special concern.

- Developing water quality quidelines for ecosystems, monitoring, policy planning and implementing are iterative processes. For Lake Balaton there were a detailed monitoring and adjustments initiated when targets were not met or they were unrealistic or there were scientific evidences of ecosystem changes. With positive alterations of the ecosystem and phosphorous loads in the early 2000s it became clear that with the strong reduction of the sewage phosphorous load it is sufficient to monitor annual average TP loads (and Chl-a) being more accurate than BAP. At the same time it was found that two additional indicators (a structural phytoplankton index (Padisák 2002) and the chironomidae biomass (Specziár 1999) contributed to refine the description of nutrient availability and the ecosystem under change.

Figure 5.6: Components of the external TP load from the Lake's basins, before and after taking control measure (late 1990s, early 2000). Source: Somlyódy (2015b).



From monitoring to policy making (Steps 6-9)

Monitoring and modelling

It is the very nature of water quality problems that the state depends on man-made pressures (nutrient loads in case of this example case study) and natural factors (air temperature, solar radiation, stream-flow rates, wind and others). In order to reach a solution it is relied upon available theoretical and empirical knowledge, as well as tools like modelling, all formulated in a solid methodological framework. Within this, monitoring plays a key role which may have various goals: trend detection, estimation of annual average loads of tributaries, model development, regulating effluent standard violations, surveillance and others (see »Sections 2.6 and 0). It is atypical that a single state variable as an indicator is monitored or modeled; the interest is rather in a number of indicators and derived quantities such as nutrient loads. Hence, an integrated monitoring network covering natural and man-influenced forcing functions need to be developed.

Here a distinction should be made between two phases: understanding and planning, and this applies to the entire decomposition-aggregation approach developed for Lake Balaton. In the course of Phase 1 historical data were used to calibrate and validate the models. Subsequently, in Phase 2 it is tried to predict the future in order to achieve the goals set. Scenario analysis is frequently used as well as optimization or multi-criteria assessment, the pre-condition of which is to get synthetic time-series generators for the two types of forcing functions. One of the difficulties can be that typical time scales of inputs may be rather different which calls for careful design and implementation of adaptive monitoring.

For Lake Balaton a number of issues related to monitoring, modelling and policy development was analysed. Next, some of the steps of the approach are discussed (Somlyódy and van Straten 1986, van Straten 1986a, Kutas and Herodek 1986, Luettich and Harleman 1986, Shanahan and Harleman 1986).

- Six alternative, dynamic four-box (Basin I to IV) nutrient cycle (load-response) models were developed and calibrated. Coupling was also made with the aggregated hydrodynamic-transport model. Calibration and validation were only partially successful: to capture the spring algal bloom and the August one in the same year required parameter adjustments for the models. In spite of it, the Chl-a = f(phosphorous load) responses were similar (and nearly linear) for all the model alternatives. This feature was demonstrated by numerical experiments when simulations were performed by systematically changing external phosphorous loads.

- Annual average nutrient loads were developed in three steps. First an estimate from routine monitoring raw data was derived. Second a refinement was made by involving pilot zone event-based observations and uncertainty analysis on the daily Zala River data to evaluate the influence of infrequent sampling. Third, cross-checking was made by evaluating nutrient source data, losses, seasonal changes in tourism, and unit areal loads.

- Daily forcing function data were aggregated to weekly, monthly and annual averages, and one of the nutrient cycle models was run to see the impact on (Chl-a)max. The exercise led to somewhat surprising outcome. Accordingly, the sensitivity on the length of the averaging period is relatively small and monthly or annual averages can be employed (it decreases with increasing residence time). This conclusion is fundamental: monitoring can be pragmatically developed such that monthly/annual average loads of individual basins are satisfactorily estimated.

- On top of all these, a simple regression model was developed for monthly phosphorous loads of the Zala River reflecting man-made impacts, natural variability and uncertainty. A similar approach was used for other smaller tributaries by utilizing statistical parameters of the flow rate for which much more data were available than for the loads. A simple autoregressive model was developed for daily water temperatures which then was cross-correlated with weekly mean global radiation (weekly to filter out huge noises). As a final step monthly averages were derived. Thus, synthetic time series generators of all the forcing functions were available to run the lake eutrophication model in a Monte Carlo fashion. Simulations were performed under various load reduction assumptions for two cases:

    I. computation of the joint impact of man-made and natural (uncontrollable) factors on water quality and
    II. analysis of the effect of uncontrollable factors alone. This showed a strikingly wide fluctuation in (Chl-a)max of Basin 1 (±40 % around the mean) which may mask the effect of considerable load reduction. The impact depends on residence time similarly as noted before.

These analyses allowed capturing experiences with monitoring, an approximately linear phosphorous load – indicator short-term response with stochastic variability, uncertainties and internal phosphorous load (the long-term impact was derived from a simple P mass balance equation describing the renewal process of the sediment (Lijklema et al. 1986).

On top of these a management optimization model was developed. This included continuous decision variables, non-linear cost functions (upgrading and P removal at WWTPs, sewage diversion, pre-reservoirs and others), transfer coefficients, an expected value - variance objective function which maximise the improvement of water quality expressed by (Chl-a). The linearized model was solved by Linear Programming (LP) (Somlyódy 1986). Another, truly stochastic model was developed as well (Somlyódy and Wets 1988). The outcome was a comprehensive sensitivity analysis, a detailed policy (including priorities, sequencing, costs and others) and above all a modelling based synthesis of many water and catchment related processes, uses and activities.

Policy making, public involvement and communication
In the early 1980s Hungary was among the European countries practicing the centrally planned economy. There were very few experiences available on how to professionally handle environmental problems. The methodology of Environmental Impact Assessment was not yet functioning. Water legislation was existing but enforcement was missing. Public involvement and open planning were still practically unknown. More or less the same applied for the application of decision sciences- Non-Governmental Organizations (NGOs) just started to come into existence. For outstanding issues, decisions were made by the Council of Ministers. This was also the case for Lake Balaton, the largest

recreational area in Hungary, the “Hungarian Sea” as it was and is often called, i.e. a national asset.

Lake Balaton and its catchment form a complex problem of several dimensions such as limnology, hydrology, hydrodynamics, meteorology, land use management, sewage treatment, economics and many others. There were many questions raised which were difficult to answer. Nitrogen or phosphorous is limiting or both of them? Which is a faster process, algae growth or sediment adsorption? How large is the internal nutrient load? Is sewage or agricultural diffuse load control more important? What are the costs? And so on.

The Lake Balaton institutions are also rather sophisticated. The lake and its region belong to the government, to several offices of ministerial level (housing and construction, agriculture, water, technology development, science), three counties, three water district authorities, and others.

At the time of launching the decision making process several expert groups were formed which discussed key scientific aspects of eutrophication and its control. Thirty to fourty technical reports were available. There were policy summaries under preparation too. Length varied between one and ten pages depending on which decision level was targeted. There was also a plan for various public hearings and consultations with leading decision makers.

As it turned out quickly, the entire process was rather unusual. The bottleneck was communication or the lack of it. Even scientists had difficulties to reach a consensus on key elements of the problem and what actions should be taken. Communication with the public or politicians was even worse. Concepts and expressions such as primary production, algae growth, Chl-a, limitation, blue-green algae, and N-fixation were not really known and their use led to confusion rather than conclusion. This was the moment when the Hungarian Academy of Sciences, perhaps the “most trusted” institution in the country started to play a leading coordinating role. Language was significantly simplified, it was suggested that professionals try to avoid the use of technical words as much as possible. Policy summaries were adjusted accordingly.

- Characterize the current situation.
- Formulate the problem and set objectives.
- Develop a feasible methodology to handle the given complex water body – catchment issue.
- Identify main natural and man-made factors, pressures and stressors.
- Create a data base by using existing information even if data are scarce.
- Start data collection as early as possible.
- Look for existing maps and remote sensing.
- Do a rapid and simple assessment.
- Prepare a concept on how to handle uncertainties, inherent features of water problems. Do a few rough estimates right at the beginning.
- Perform a time scale analysis for interlinked processes (e.g. nutrient loads and trophic state) and draw conclusions for their sampling.
- Prepare a sampling strategy/monitoring plan. Use simple statistics to gain information efficiently.
- At each step check mass balances. The process of planning and implementation is consisting of cycles and iterative steps, largely determined by the outcome of monitoring.
- Proceed from simple towards complex.
- Adaptation and integration are the key features at all the levels.
- Do not forget post-monitoring.

Box 5.2: Summary of the most important steps.

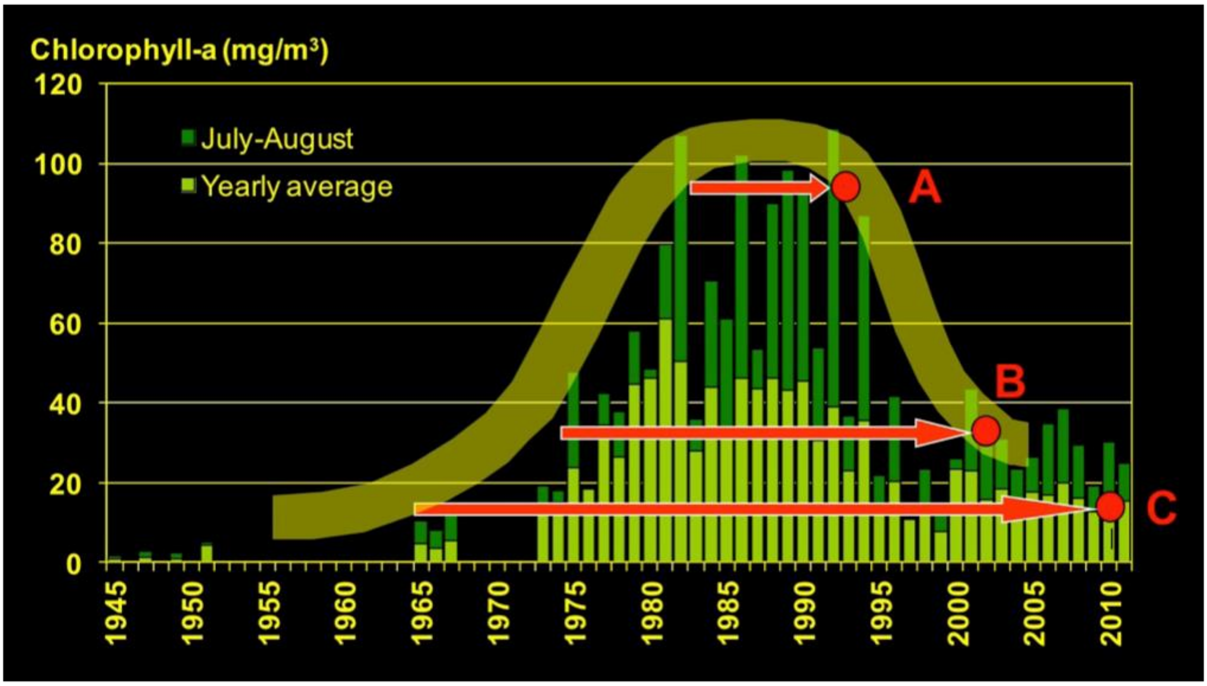
As a key element of the modified communication strategy, the presentation of the three targets specified was changed completely. It was recognized that people still remembered the colour of the water in the early 1970s and even in the early 1960s, and could compare them to the current colour. The reaction of politicians and journalists were the same. The targets specified became clear to them, they were easy to understand and memorize, particularly when they were called as A, B and C meaning A – no further deterioration, B and C – reaching the water quality of the early seventies and early sixties, respectively (see »Box 5.1). A smooth process was realised afterwards which led to the acceptance of measures as proposed (see above), together with setting priorities and allocating financial resources in the state's budget. Box 5.2 summarizes the most important steps with key hints. This sequence of activities reflects well the 9 steps proposed in the Framework.

Closing remarks

What happened afterwards? Till 1986 the implementation proceeded more or less as planned. This was followed by economic recession and political change leading to tremendous alterations in all aspects of life and to a slow down of the remediation program. But, due to increasing costs, fertilizer application was reduced by a factor of 4 or 5 and thus diffuse loads strongly decreased as well (it was not planned). Droughts at the early 2000s also contributed. As of today practically all the measures listed earlier have been implemented, sewage loads has nearly disappeared and the external phosphorous load was cut by about 50 %. Trophic state did not change until 2004 when a sudden improvement happened (»Figure 5.6 and see also »Figure 5.2). The likely reason was the fast renewal of the sediment and hence the reduction of the internal phosphorous loads. Simultaneously, the structure of phytoplankton also changed: now blue-green algae have only a small fraction in the total biomass (»Figure 5.7).

Today the lake is close to the status C, but not yet reached this desired goal. Obviously, the implementation of the strategy has not been completed. Fertiliser application seems to increase again. To reach the oligotrophic state in the entire lake requires further load reductions of increasing unit costs. Additionally, new problems emerge continuously (such as climate change) of possibly surprising impacts. Thus, the advice is to continue the observations and measures as defined by the 2003 revision of the Action Plan (Somlyódy et al. 2003).

Figure 5.7: Algae biomass: the impact of load reduction.
Lake Balaton, Keszthely Basin (Basin I, yearly and July – August average Chl-a).
Source: Kovacs and Clement (2009)



Example II: The Upper Tietê River Program, Brazil

Introduction

The ongoing rehabilitation programme to improve the water quality status of the Upper Tietê River in the state of São Paulo in Brazil is a good example of how long and how expensive concentrated and coordinated actions are needed to restore water quality to an acceptable level in rivers and reservoirs after decades of pollution caused by growth of population and increasing economic activities. The Tietê River has a WQG established in 1977. In the São Paulo Metropolitan Region (SPMR) the river is aimed to reach is a river class IV, with the goal to have at least 2 mg/l of Dissolved Oxygen (DO) in it, among other parameters. The short presentation of this successful and still ongoing cleanup program will prove again that the 4 phases- 9 steps approach as outlined in »Section 4.1.4 can very well serve as a comprehensive framework for present and future water quality improvement programs and long term adaptive management of water quality.

The Upper Tietê River and its basin

The Tietê River is a tributary of the Paraná River. It forms part of the Plata Basin. The Tietê River has a total length of 1,136 km, a basin area of 71,938 km² and is completely included in São Paulo State, southeastern Brazil as shown in »Figure 5.8.

The present case study focuses on the upper part of the Tietê River. It has a length of 243 km. Its headwater is located at 1,115 m above sea level. The Upper Tietê River Basin has a drainage area of 5,720 km².Urban areas occupy 37% of this basin. This is a unique case of a strongly urbanized catchment which is located at the uppermost part of a river basin as shown in »Figure 5.8.

Figure 5.8: Tietê River Basin. Source: FUSP (2009).

The Upper Tietê River flows through the SPMR, the largest urban and industrial concentration in South America which includes the city of São Paulo and 38 adjacent municipalities. 19% of the entire Brazilian Gross Domestic Product (GDP) and 50% of that São Paulo State is generated within the Upper Tietê Basin. The basin has 20 million inhabitants (as of 2010) with a population density of 3,496 inhabitants per km².

Human water consumption is larger than the water available in the basin, and a large portion of water (31 m³/s) has to be imported from neighboring basins. It is estimated that 11 m³/s (17% of the total water supply demand) is extracted from groundwater. About 7,000 wells in the basin contribute to cover the water demand.

Due to the dense population, the water supply also depends on a series of reservoirs upstream of the urban agglomeration. The main reservoirs in the Upper Tietê Basin are Billings, Guarapiranga, Ponte Nova, Jundiai and Biritiba-Mirim.

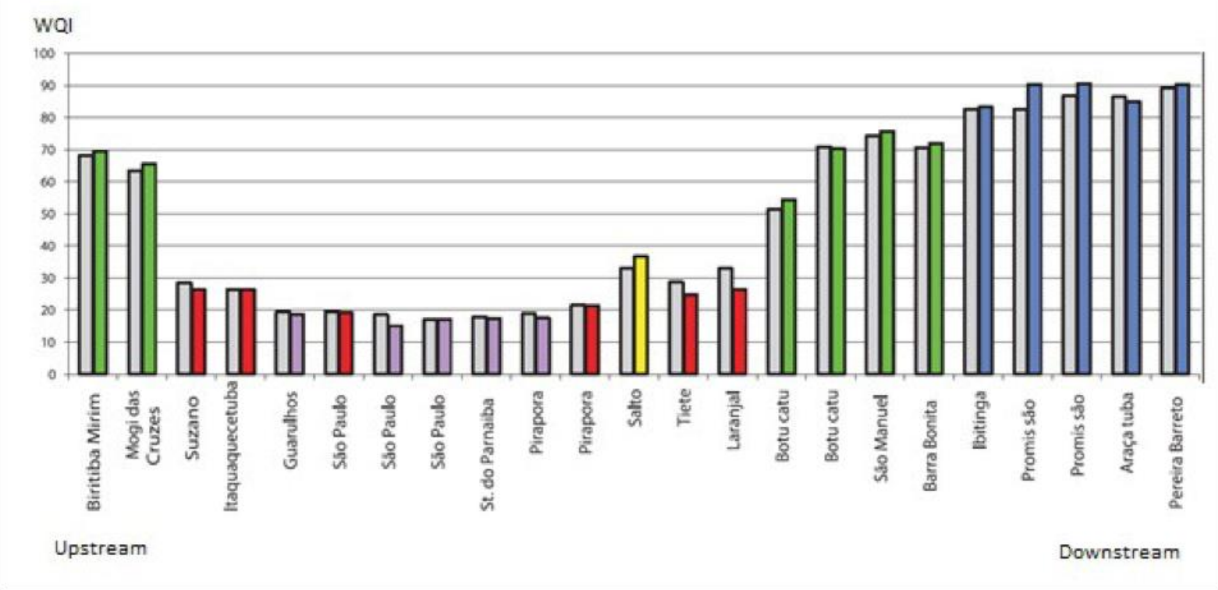
The annual average precipitation is 1,400 mm, with even higher values (2,500 mm) on the southern rim of the basin. Due to manifold human interventions the hydrology of the basin is very complex. Structures for flood control, water supply and flow regularization have changed the natural flow regime (Devkota and Imberger, 2012). The average monthly flows of the Tietê River in its upper basin varies between 100 and 240 m³/s.

The Upper Tietê River is one of the most polluted water courses in Brazil. The lack of adequate treatment of domestic and industrial wastewater in SPMR caused serious deterioration of the quality of the river water. High concentrations of Biological Oxygen Demand (BOD), phosphorus, ammonia-nitrogen, pathogenic microorganisms (E.coli, Shiguella flexneri, and S.boydii) and toxic chemicals are observed, indicating the strong contribution of domestic and industrial waste water (CETESB, 2014; Abraham et al, 2007).

The Water Quality Index (WQI), which is used in Brazil to assess the water quality status of water bodies (ANA, 2012) shows along the Upper Tietê River the impact of pollution emanating from SPMR. The headwaters located near the city of Biritiba-Mirim still have a good water quality, but as the river enters the SPMR (between the cities Suzano and Pirapora) a strong decline of water quality can be observed. Downstream of Pirapora a recovery of water quality occurs and the Upper Tietê River has again good quality in its final reach (»Figure 5.9).

Figure 3.4.3: Water Quality Index along the Tietê River in 2013 (coloured bars) and the average between 2008-2012 (grey bars).

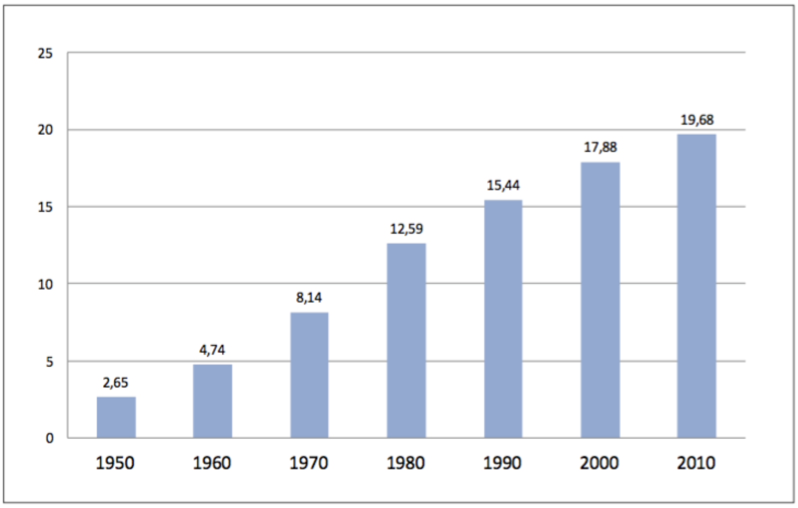
Source: CETESB (2014).



The uncontrolled population growth of SPMR (see »Figure 5.10) around two major reservoirs (Billings and Guarapiranga) has impacted the water quality because of the release of sewage, garbage and diffuse pollution. These led to the eutrophication of the reservoirs and the occurrence of unpleasant taste and smells in the drinking water supplied to the population. These reservoirs are important for the water supply of the SPMR. They supply 19 m³/s water covering the domestic water demand of 5.4 million people. They are classified at present as eutrophic- water bodies (see also »Figure 5.11).

Figure 5.10: Population of São Paulo Metropolitan Region (million inhabitants).

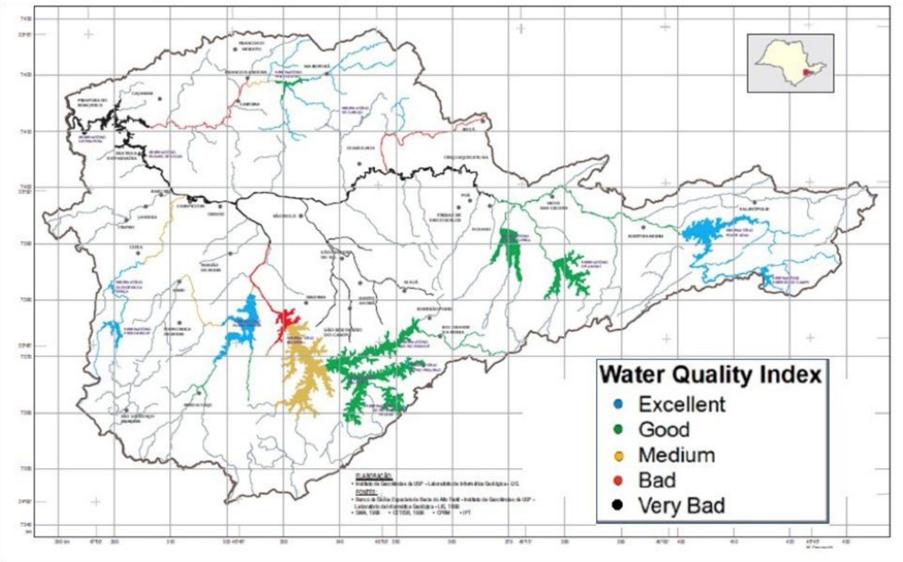
Source: IBGE (2012).



Cyanobacteria blooms and three variants of microcystin were observed in some reservoirs in the Upper Tietê River Basin (Moschini-Carlos et al, 2009). High concentration of toxic chemicals (e.g. 1.1.-dichloroethene, chlorophorm, methylene chloride and toluene) are also observed in the Upper Tietê River Basin. Among the metals, lead concentrations are high, particularly in the Tietê River (maximum of 0.15 µg/liter), chromium concentrations in the Pinheiros River, a tributary of the Tietê (maximum of 0.31 µg/liter for both, total and trivalent chromium). Soluble iron concentrations are higher in the Pinheiros River compared to the Tietê River (maximum of 14.3 mg/liter versus 3.8 mg/liter) (Cunha et al, 2011).

Figure 5.11: Upper Tietê River - Water Quality Index.

Source: FUSP (2002).



Cyanobacteria blooms and three variants of microcystin were observed in some reservoirs in the Upper Tietê River Basin (Moschini-Carlos et al, 2009). High concentration of toxic chemicals (e.g. 1.1.-dichloroethene, chlorophorm, methylene chloride and toluene) are also observed in the Upper Tietê River Basin. Among the metals, lead concentrations are high, particularly in the Tietê River (maximum of 0.15 µg/liter), chromium concentrations in the Pinheiros River, a tributary of the Tietê (maximum of 0.31 µg/liter for both, total and trivalent chromium). Soluble iron concentrations are higher in the Pinheiros River compared to the Tietê River (maximum of 14.3 mg/liter versus 3.8 mg/liter) (Cunha et al, 2011).

Groundwater contamination is caused mainly by the leaks of gasoline from gas stations and industries. About 70% of the contamination episodes recorded was caused by aromatic solvents and liquid fuels. Nitrate and microbiological contamination of groundwater is found mainly on the periphery of SPMR without sewage collection systems (FUSP, 2009).

The contamination of the Tietê River caused a strong deterioration of environmental conditions. Fish surveys carried out along the Tietê River showed significant reduction of fish diversity as the river enters the urban areas. Fish is completely absent when the river flows through the city of São Paulo. Fisheries in the Upper Tietê Basin are now restricted to upstream reservoirs that are less affected by pollution (Barrela and Petrere, 2003).

Irrigation with polluted water is also a concern in areas around SPMR where mainly vegetables are cultivated. Pinheiros River is a large urban breeding area for mosquitos. Insecticides are applied along the river to eliminate the mosquitos and other disease vectors. Organophosphate compounds were found in the water of Pinheiros River (Cunha et al, 2011).

Downstream of the SPMR water pollution causes excessive foams by detergents. The accumulation of debris and the spread of unpleasant odors causing respiratory problems in the city of Pirapora.

The total domestic organic load in the basin is 635 tons of BOD/day. The industrial organic load is 26.4 tons of BOD/day and the industrial inorganic load is 307 kg BOD/day (FUSP, 2009). Over the last 6 years it is estimated that between 190 and 494 tons of BOD were discharged each day in the outflow of the Upper Tietê River at Pirapora. As the consequence the section of the Tietê River that runs through the SPMR frequently has low levels of DO (below 2 mg/liter) and in many tributaries (Pinheiros, Tamanduatei) the values are close to 0 mg/liter.

The urban core of the SPMR shows the heat island syndrome that causes the formation of convective storms of high intensity and spatial concentration, causing floods on small urban watersheds and extensive stormwater runoff thus contributing to the significant pollution load from diffuse sources.

Climate change is an important factor of water quality of the Upper Tietê River

Basin. An increase of rainfall can result in more diffuse pollution, while a possible decrease of rainfall can reduce the capacity of streams to assimilate pollutants.

The above summary reflects the physical, chemical and biological status quo of the basin with a short characterization of the history of the responsible processes. It became obvious that the impaired state of freshwater ecosystems ultimately undermine their ability to provide services for human use and utilization. This part of the case study can be identified as Steps 3 and 4 of the proposed Framework approach (see Section »4.1.5, »0 and »4.4). Step 2, the classification of inland water ecosystems is covered in the above summary as the “Cleanup Program of the Upper Tietê River” is implemented to fit class IV of the WQG established 1977. As far as the assessment ofwater quality is concerned the WQI (CETESB 2013) method is used.

The initiation of the Upper Tietê River Cleanup Program

Until the 1950s several rivers in the SPMR maintained good water quality conditions. The event “Swim Crossing of São Paulo” was held on the Tietê River until 1944. Fishing and other water sports were important activities. However since the 1950s the high pollution levels in the rivers have prevented these activities. There are still people around who can remember the state of the river prior to the extreme pollution phase. These remembrance factors, like in the case study of Lake Balaton (see »Section 5.2) can be very important to formulate a vision (Step 1 of the Framework) to be achieved. Aiming to bring back a water quality and ecological status people can recall and associate with is essential to secure public support for policy as shown also in the case of Lake Balaton (see »Section 5.2).

The unfolding of the Upper Tietê River Basin Cleanup Program, its duration and width including institutional and financial provisions clearly underline the close relationship between Steps 8, 9 and 1 in the Framework. As the “turning point” of the proposed adaptive cycle involves these steps, it is frequently needed that some elements of Steps 8 and 9 are implemented in parallel or even ahead of Step 1 and thus would precede to address problems under Steps 4-7.

In the late 1980s NGOs and the media played an important role in a campaign to clean up the river. In 1991 a petition calling on the state government to clean up the Tietê River was signed by 1.2 million people. In the same year the São Paulo State Government launched the Tietê River Cleanup Program. The program is funded by loans from the InterAmerican Development Bank, the Brazilian Development Bank (BNDES) and by resources of São Paulo State Government. The program includes the construction of new WWT facilities, expansion of existing plants and construction of sewage collection networks and interceptors. These measures are clearly aiming to reduce the stressors the freshwater bodies are exposed to.

The program was divided in three phases with a total investment of US$ 2.65 billion. The third phase was completed in 2015. By then, the collection of waste water was increased from 70% in 1991 to 87% and the treatment of domestic waste water will increase from 24% to 84%. The progress of the Tietê River Cleanup Program will be presented in »Section 5.3.4.

As far as the water governance and water management (Steps 8 and 9 in the Framework) of the Tietê Basin is concerned there are several agencies involved. In Brazil the domestic sewage collection and treatment is a mandate of municipalities. In the Tietê basin, most municipalities transferred sanitation services to the state sanitation company, but some maintained their own municipal sanitation services. Therefore, not all municipalities participated in the program developed by the State Sanitation Agency, which caused delays in implementing the vision of a clean Upper Tietê River.

Water quality is monitored by the São Paulo Environmental Agency (CETESB). This agency has also the mandate for water pollution control. The São Paulo State Sanitation Company is in charge of the domestic water supply and WWT at most of the municipalities of the basin. Water quantity is monitored by the State

Water Department, which is also responsible for giving the permissions for water use. The Upper Tietê River Basin Committee was created in 1991 and its main responsibility is the implementation of the River Basin Plan.

The civil society, NGOs and the media played and play an important role in the formulation and implementation of the Tietê River Cleanup Program.

Implementation of the Tietê River Cleanup Program

The first three phases of the Cleanup Program until the end of 2015 covered a quarter of a century and consumed the expenditure of 2.65 billion US$ (see »Table 5.5). The fourth phase of the program is being prepared.

Table 5.5: Phases of the Tietê River Cleanup Program.
Source: São Paulo State Sanitation Company (SABESP, 2016).

Phase	Period	Investment (US$)
First	1992 - 1998	1,1 billion
Second	2000 – 2008	500 million
Third	2009 – 2015	1,05 billion
Fourth (under negotiation)	2016 – 2025	1,9 billion
Total	**1992 – 2025**	**4,55 billion**

While the achievements of organic and inorganic load reductions are significant the fact that a new, decade long phase is under negotiations indicates that the rehabilitation of such massively polluted river system would take decades and several billions of US$.

In the process to expand the collection of sewage water the program implementation encountered a social problem which may be faced in similar project contexts worldwide. Some households in poor regions were unconnected to the sewer system because people were unable to pay for this service. In these cases the connection costs were taken over by the state government. This is an ecosystem rehabilitation investment in the sense of Step 9 of the Framework.

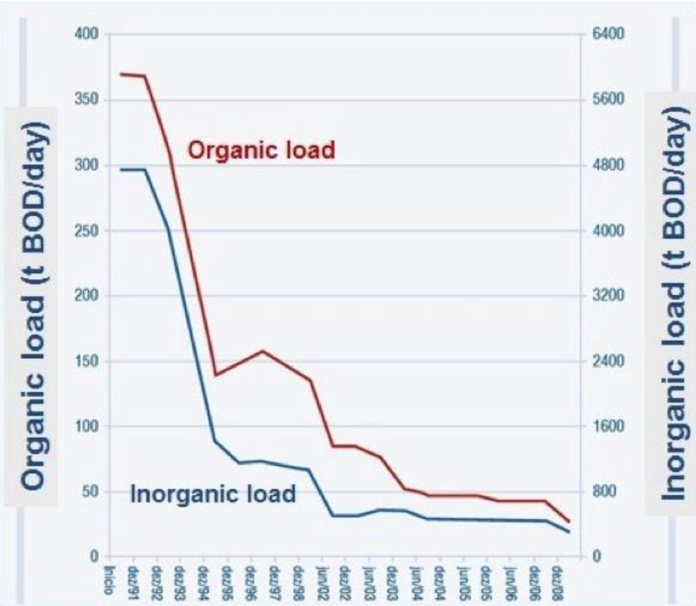
In regard to industrial pollution in 1992 CETESB identified 1,250 industries that were responsible for 90% of the industrial pollution load released into the aqueous phase. Industries were requested to submit plans and a schedule for implementing treatment systems and were supported by loans from the World Bank and the BNDES (Helmer and Hespanhol, 1997). The fiscalization of these plans and application of fines between 1992 and 2008 led, as shown on »Figure 5.12 to a 93% reduction of industrial organic load and a 94% reduction of the industrial inorganic load (CETESB, 2008).

The main result of the increase of domestic sewage treatment and control of industrial pollution was the decrease of the river pollution downstream of the SPMR. In 1992 a total length of 260 km was affected by heavy pollution downstream of the Upper Tietê River Basin. By 2014 this length was reduced to 100 km. This contributed to the return of higher order aquatic life (fishes) to some locations and the reduction of unpleasant odors.

The condition of the Tietê River within the SPMR is still very critical. Even after the completion of Phase 3 of the Cleanup Program by 2016 the water quality will not comply with requirements of human uses like recreation and fisheries (Davis and Hirij, 2003).

Figure 5.12: Reduction of organic and inorganic industrial loads in the São Paulo Metropolitan Region.

Source: CETESB: 2008.



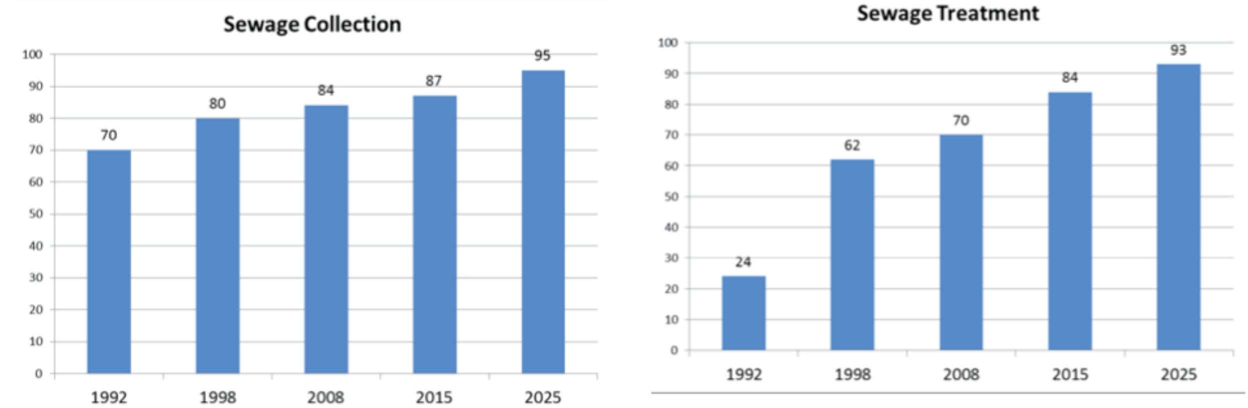
The recovery of the Tietê River is a long process that will need continuous investments over the next years. Since 1992 the Tietê River Cleanup Program kept going through changing governments. This is the biggest river cleanup project in Brazil and shows the importance of public participation and the creation of sustainable governance and financing mechanisms, thus that of Steps 7, 8 and 9 of the proposed Framework.

Achievements and outlook of the Tietê River Cleanup Program

»Figure 5.13 shows the impressive achievements in the area of sewage treatment. By 2015 the percentage of treated sewage more than tripled, while sewage collection grew from 70 to 87%. Irrespective of these increases there is not yet significant improvement which city inhabitants would be able to notice.

Figure 5.13: Evolution of sewage collection and treatment in the São Paulo Metropolitan Region.

Source: São Paulo State Sanitation Company (SABESP, 2016).



The fact that the improvement of freshwater ecosystems is slow and may not take place visibly for those who fought for it and ultimately footing the bill of the restoration, highlight the need for multi stakeholder involvement, effective public communication, reporting and information dissemination over a considerable timespan.

Results, like these are powerful means to convince people about wide scale benefits, even direct human benefits and also the urgency of rehabilitation of freshwater ecosystems.

After the 4th phase of the Tietê River Cleanup Program in 2025 it is envisaged that the entire river should return to a solidly aerobic state with significant increase of downstream reaches accommodating again higher value aquatic life (the upstream reaches already have a good quality and will not be affected by the program). Yet, it has to be acknowledged that even by 2025 the "urban part" of the river possibly will not reach a status close to a biologically active habitat. Given the enormous land use pressure and corresponding level of stressors this is unlikely to be a viable state to be aspired for. However the basin wide cleanup will ultimately contribute to the reestablishment of high value freshwater bodies downstream of the São Paulo Metropolitan Region (SPMR).

- While the case of the Upper Tietê River and São Paulo, the fifth largest urban agglomeration on earth is exceptional, there are still important transferable lessons to be learnt like public participation (Step 1 but also 7, 8 and 9 of the Framework)
- Continuity of public policies (which highlights the importance of a well - conceived Step 9)
- Support of poor households and communities (meaning innovative, socially balanced funding schemes) (Step 9)
- Importance of water quality monitoring and communication (Steps 6 and 7 of the Framework, but ultimately also well conceived guidelines (Step 5) are implied).

The Upper Tietê Basin case study has shown the viability of the proposed 4 phases - 9 steps approach as presented in »Section 4.1.5 as a general and flexible framework adaptable for water quality rehabilitation, monitoring and management. The case of the Upper Tietê River proves that even if political dedication, good governance, public interest and involvement, funding and able technical services are available the rehabilitation of deteriorated freshwater ecosystems will remain tasks for decades. This should serve as the most powerful message to find sustainable development pathways rather than following the ultimately infeasible "impairing then repairing" (Vörösmarty, 2013) cycle which hitherto overwhelmingly characterized industrial and urban development since the second half of the 19th century.

Example III: Mining impacts on the Strickland River, Papua New Guinea, and the management policy adopted

Introduction: the context of the example

In many developing countries, water quality monitoring programs are unlikely to have been developed by government agencies at a broad national or even regional level. In the absence of field data and with limited technical capacity and financial resources, screening of pressures/stressors and risks may provide the only means of initial estimates of freshwater ecosystem health (see Step 4.3). However, detailed water quality information may be available from the private sector, for example, mining companies operating within river basins, who are often required to assess the potential impacts of their operation on the downstream ecosystems and communities.

In this example a review of a monitoring program and development of an ecosystem health report card for the Strickland River in Papua New Guinea is presented (»Figure 5.14). The Strickland River arises in the Western Highlands of New Guinea (2000-3000m asl) with an annual rainfall of ~3600mm and joins the Herbert River, which drains Lake Murray, and then the Fly River before entering the Gulf of Papua. The high rainfall in the upper catchment, steep and erodible terrain and occasional seismic activity make the region prone to landslides and avalanches, contributing large amounts of sediment and debris into the river system. Porgera Joint Venture (PJV) established a gold mine in the upper Porgera valley in the late 1980s, and began gold production 1990 initally processing 1500 t/day of ore and expanding to over 16,000 t/day by 1996 (CSIRO 1996). Approval of the mine was on condition of the establishment of an Environmental Management and Monitoring Program and an initial baseline sampling was undertaken in 1989. PJV has undertaken an extensive environmental monitoring program of the Strickland River system since 1990, with details on water quality, sediments and biota presented in the PJV Annual Environment Reports (e.g. http://www.barrick.com/files/porgera/2012-Porgera-Annual-Environmental-Report.pdf).

The monitoring program was independently reviewed by CSIRO in 1996, and again by an independent team ten years later (Bunn et al. 2006). The monitoring data were subsequently analysed and presented as a report card on river health in 2009 (http://www.barrick.com/files/porgera/PEAK-Porgera-Report-Card-2010.pdf).

Setting objectives (Step 1)

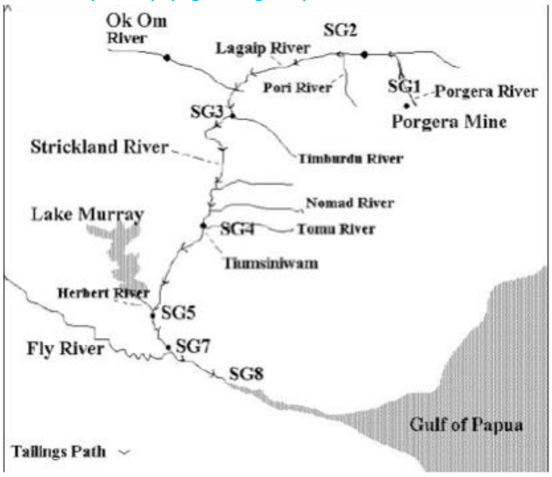
The rationale for the proposed environmental monitoring program was initially described as: "A sound program of environmental management and monitoring will enable the PJV to minimise the impacts of the Porgera Gold Project on the environment, to continually assess the significance of impacts and take remedial action where unacceptable impacts are indicated. The program is designed to provide the PJV with pertinent information regarding the effects of its mining operations on the environment, as well as satisfying environmental quality objectives set by the PNG Government." The program of Papua New Guinea was focused on the impacts of mine-derived contaminants, with an emphasis on compliance monitoring at point SG3 (monitoring site) (»Figure 5.14).

Bunn et al. (2006) noted that, although the objectives of the program were well understood by the PJV Environment Department, they needed to be formally stated in a public document. They recommended that the overall objectives of the monitoring program be included in future annual environment reports, and an overview document be prepared that describes the objectives of the each component of the monitoring program and the rationale for inclusion of particular indicators and their sampling design.

Figure 5.14: Location of the Strickland River in Papua New Guinea and the Porgera Gold Mine.

SG1 to SG8 represent monitoring sites (right Figure).

Source: PJV (2003) (left Figure); IIED and WBCSD (2002) (right Figure).



Classification of the river ecosystem (Step 2)

The review by Bunn et al. (2006) acknowledged the marked differences in biophysical processes between the headwater streams, confined river gorge, floodplaine river and associated wetlands and lakes of the Strickland River system. Based on topography, discharge and water body type, the environmental monitoring sites were grouped into three distinct zones for analysis:

vi. the Upper catchment, including the compliance site at SG3;

vii. the Lower catchment (down to the Fly River confluence) and

viii. Lake Murray.

'Reference' river sites included tributary systems upstream of the Porgera mine or the confluence with mine-affected Porgera River. Data for Lake Murray were reported for three regions, related to distance from inflows from the Strickland River, with the northern region (furthest from the river) considered as reference (Bunn et al., 2010). Sampling methods and some indicators differed between these water body types, and it was necessary to account for natural differences in some water quality (e.g. turbidity) and biotic (e.g. fish diversity) parameters when establishing benchmarks.

Setting the basin-scale context and desktop screening (Steps 3 and 4)

Mining represents one of the most significant economic activities in Papua New Guinea and also represents one of the significant threats to river systems. A country scale assessment could be undertaken to identify priority river basins of concern and, in most of these, detailed monitoring programs are likely to be in place, funded by the mining companies and overseen by the Government. A considerable body of information on the Strickland River was available on riverine inputs from mining operations and processing in CSIRO (1996) review, and the annual Environmental Monitoring Annual Reports[62].

Establish water quality guideline values for indicators of concern (Step 5)

Long-term (since 1990) data from the PJV Environmental Monitoring Program were collated for the preparation of a 2009 Report Card, and five groups of indicators identified:

- Dissolved metals (arsenic, cadmium, copper, lead, mercury, nickel, silver and zinc)
- Metals in sediments (arsenic, cadmium, copper, lead, mercury, nickel, silver and zinc)
- General water quality parameters (conductivity, total suspended solids and pH)
- Metals in fish tissues (arsenic, cadmium, copper, lead, mercury, nickel and zinc)
- Fish composition [Biomass (B); Average biomass per individual (B/I), Biomass proportion of top predators (trophic group 1), Biomass proportion of aquatic invertivores (trophic group 2), Biomass proportion of terrestrial insectivores (trophic group 3); Biomass ratio of top predators: detritivores (B1:4), (trophic group 4 = detritivores)].

Long-term data from reference sites were used to calculate reference values for each indicator. Reporting was based around a system of trigger levels of concern for ecosystem health. The median value for a particular measure (e.g. dissolved arsenic concentration) was calculated for each site in 2008 to assess its ecosystem health in relation to a pre-determined reference value (RV) and two trigger levels of concern: an Early Trigger (ET) value and a 'true' Trigger Value (TV) (Bunn et al., 2010).

[62] e.g, http://www.barrick.com/files/porgera/2012-Porgera-Annual-Environmental-Report.pdf

A conservative measure of reference site condition within each region (e.g. the 80th percentile of dissolved arsenic concentration in upper reference sites from the start of data collection to the present) was used as the Reference Value (RV). This represented an acceptable level of health, which equates to a low level of concern. For most dissolved metals and other water quality parameters, TVs and ET values were taken, respectively, as the 90th and 95th percentile Australian and New Zealand guideline trigger values for ecosystem protection (corrected for hardness of 90 mg L-1 CaCO3). For sediment metals, TVs and ET values were taken, respectively, as the Interim Sediment Quality Guidelines (ISQG) ISQG-Low and -High Australian and New Zealand guideline TVs for ecosystem protection.

In the absence of any documented values of concern, TVs for metals in fish tissue were set at three times the reference value. Similarly, trigger values for biotic measures were arbitrarily set at one third of the reference value.

The report card covers the riverine system from just downstream of the Porgera mine (Porgera River) to Lake Murray on the Strickland River floodplain.
Porgera Joint Venture (PJV) discharged approximately 6.05 million tonnes of of tailings in 2008 to the downstream riverine systems (Porgera, Lagaip and Strickland Rivers). Additionally, an estimated 12.5 million tonnes of suspended sediment entered the riverine system from the erodible water dumps (Anawe and Anjolek).
The Porgera River joins the upper Lagaip and the flows west to join the Ok Om and then south down the Strickland River. There is significant dilution of the mine inputs along the way. The river flow at SG2 (Stream Gauging Station No.2) on the Lagaip contributes only a third of the flow at the compliance monitoring site at SG3 and the remainder comes from Ok Om and other tributaries (e.g. Pori and Tumbudu Rivers). It is estimated that about 20% of the sediment at SG3 is mine-derived. Only about one quarter of the river flow at SG4 is from the catchment upstream of SG3.
For the Upper River, monitoring data were obtained from sites downstream of the mine (SG1, SG2, SG3, Wankipe) and from nearby reference sites (Upper Lagaip, Pori River, Kuru River, Ok Om). For the Lower River, data from the main Strickland River (SG4, SG5, Bebelubi) and two reference systems (Baia and Tomu Rivers) were used. Data from up to 25 sites in Lake Murray were combined for three regions, according to their proximity to the Herbert River inflows: Southern Lake, Middle Lake and Northern Lake. Inflows from the Herbert River into Lake Murray occur about 15% of the time at times of high flow in the Strickland River due to high rainfalls in the mountains.
Total rainfall for 2008 at the mine site was 4.15 m, 12% higher than the long-term average of 3.72 m (since 1974). Average daily river flows in 2008 were also higher than the long term average (e.g. 904 m^3/s compared with 746 m^3/s at SG3).
Source: http://www.barrick.com/files/porgera/PEAK-Porgera-Report-Card-2010.pdf

Box 3 ‘No net loss’ policies for wetland extent

Monitoring , data management and synthesis (Step 6)

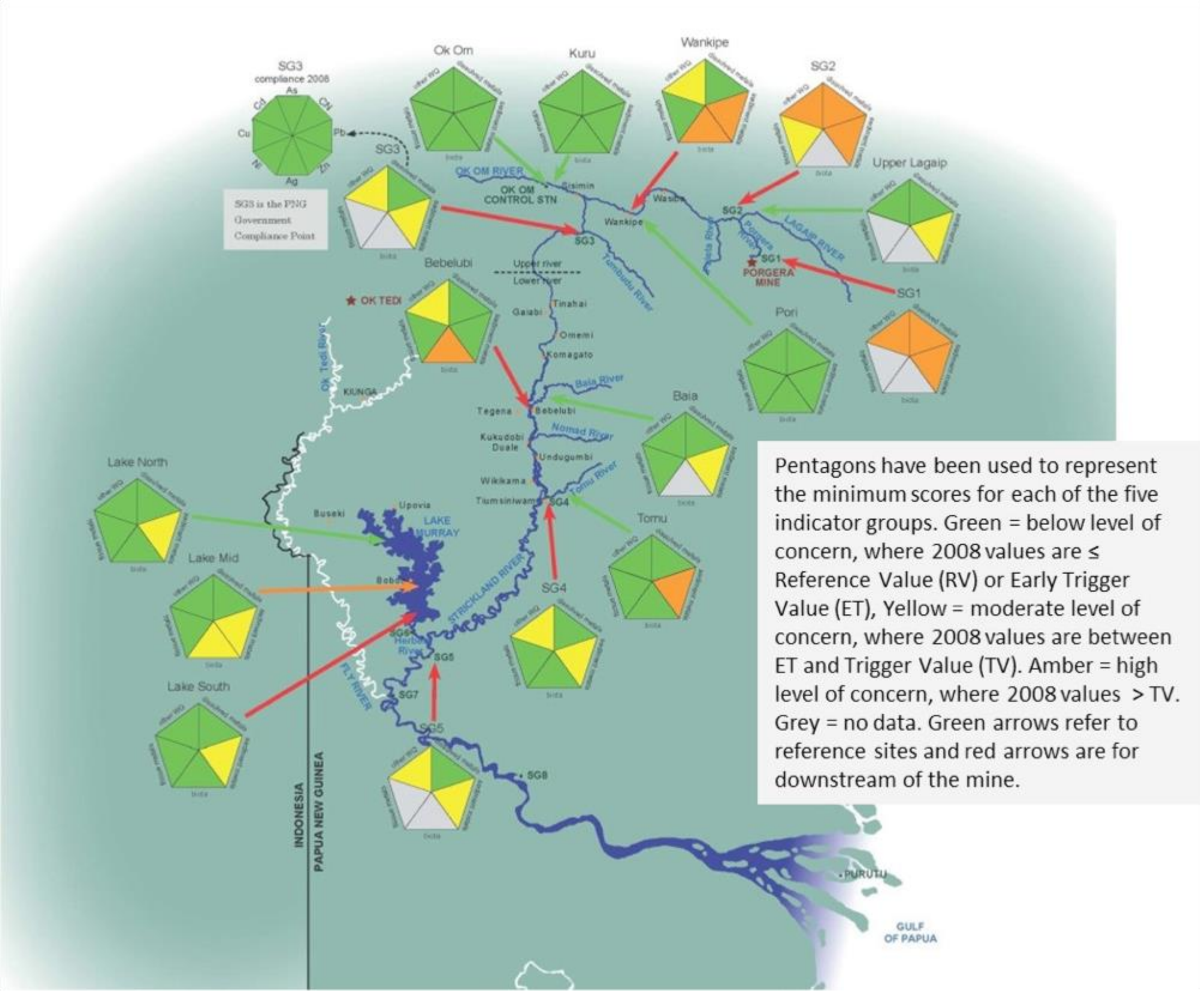
Any sites that had median values falling within the relevant RV range during 2008 were reported as GREEN (= low level of concern). Trigger levels of concern were calculated differently depending on the area of reporting, but were based on the above guideline values or indices of health in the scientific literature. An ET value was used to indicate a moderate level of concern, reported as YELLOW for sites in which the 2008 median value fell somewhere between the ET and the true TV. Sites that had median values outside the TV were regarded as having a high level of concern and were reported as AMBER.

In the case of dissolved metals, sediment metals and fish tissue metals, the overall score for the indicator group reflected the worst case score for any individual metal. For example, if a site ‘scored’ GREEN for dissolved arsenic but AMBER for dissolved cadmium, the overall score for dissolved metals would be AMBER.

In summary, levels of concern were generally determined as follows:

LOW (Green) = median 2008 value is less than or equal to RV or ET;
MODERATE (Yellow) = median 2008 value is between ET and TV;
HIGH (Amber) = median 2008 value is greater than TV.

Figure 5.15: 2009 Report card on the Strickland River, Papua New Guinea.
Source: http://www.barrick.com/files/porgera/PEAK-Porgera-Report-Card-2010.pdf



Evaluation, category assignment, reporting and communication (Step 7)

Although various indices within each indicator group were combined to a single score (see »Section 5.4.6), it was agreed that data for each of the indicator groups would be presented in the Report Card, rather than attempting to combine them into a single site score. This avoided the concerns about relative weighting of indicator groups and provided more diagnostic information as to any specific problems at each site. Data were presented as a pentagon for the five indicator groups, as described in »Section 5.4.5 (Step 5) and the report card launched publicly and made available on the company website (see »Box 5.3 and »Figure 5.15).

Governance, adaptive management and funding (Steps 8 and 9)

Although environmental monitoring data continue to be presented in annual reports (note the latest available is from 2012[63]), not subsequent Report Cards have been prepared for the Strickland River. Although there were several recommendations made by Bunn et al. (2006) to improve the monitoring program for the Strickland River, it appears that it remains largely unchanged. It would be possible to update the Report Card with more recent data and, given the long term sampling program for many of the indicators, also prepare a report on

[63] http://www.barrick.com/files/porgera/2012-Porgera-Annual-Environmental-Report.pdf

temporal patterns.

A similar synthesis of environmental monitoring data and preparation of a report card has recently been undertaken for the Watut River in northern, Papua New Guinea . This project includes a capacity building and training component with the aim that the regional university will continue to develop the Report Cards in subsequent years.

Example IV: A comparative review of a series of biological assessment programs from across the continental United States

Background

The state of the art of the WQG development in the US is discussed in some detail in »Chapter 3. Here in »Section 5.5, a particular aspect, the comparative review of the biological assessment programmes will be introduced. In 2011 the United Stated Environmental Protection Agency (US EPA) released A Primer on Using Biological Assessments to Support Water Quality Management (US EPA 2011b). This document serves as a primer on the role of biological assessments in a variety of water quality management program applications, including reporting on the condition of the aquatic biota, establishing biological criteria, and assessing the effectiveness of pollutant source controls. The document provides a discussion of technical tools and approaches for developing strong biological assessment programs and includes 17 examples of successful application of those tools. These case studies were reviewed for their value in demonstrating the Framework step-by-step approach. Results of this review have been summarized in this »Section 5.5. The entire document, including the full case studies, can be downloaded from the internet .Introduction to the examples (modified from US EPA 2011b).

Biological assessments, in conjunction with other data (chemical, toxicity, physical, landscape), provide water quality management programs the data and information necessary to document the effectiveness of management actions to protect and restore water quality and to clearly communicate that information to the public. Biological assessment data, effluent toxicity test results, physical and chemical monitoring are used to build the relationship between the stressors being managed and the biological impact of the stressors. By understanding the relationship between stressors and biological condition, appropriate management activities can be implemented towards the desired environmental improvements. The ultimate goal is a water quality management program that integrates biological, physical, and chemical data to create a more complete picture of resource conditions that supports effective implementation of management programs.

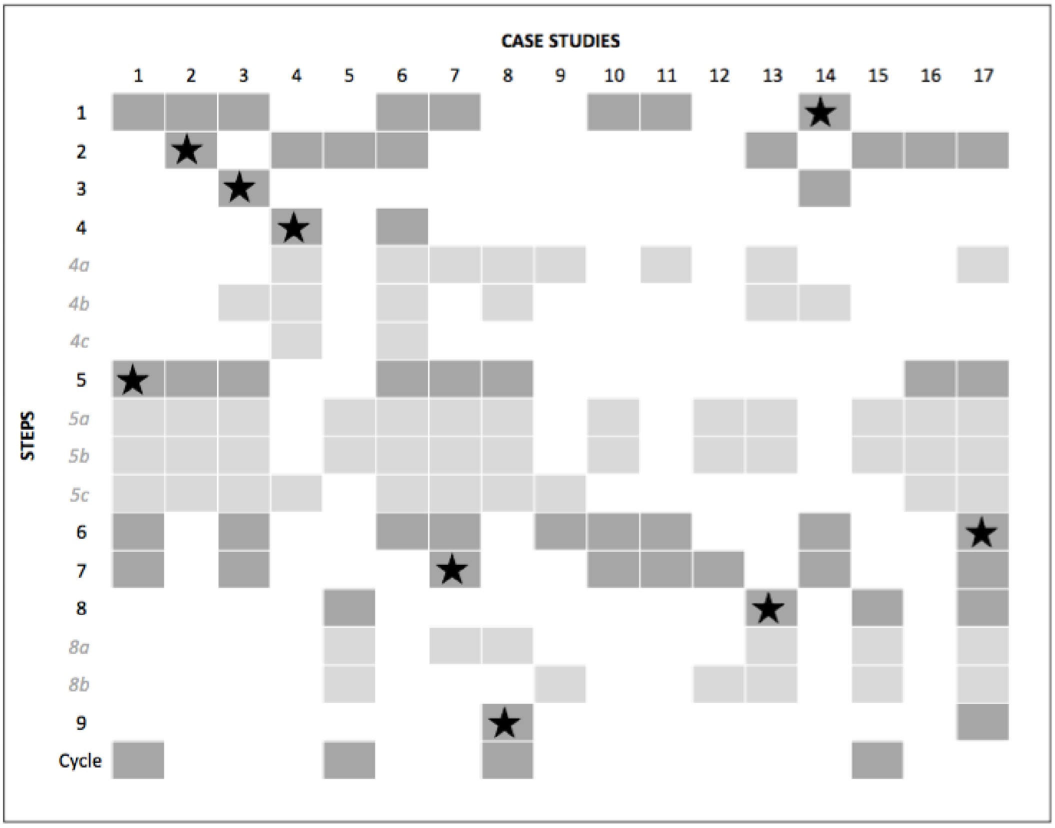
By quantifying the stressor-response relationships, it is possible to explain to stakeholders the effects of stressors on aquatic life. For example, biological assessment data can be used to document the effects on aquatic life from an undetected toxic effluent from a point source, increasing impervious surfaces in a watershed, the loss of wetlands, or the effects of channelization. Once management actions are implemented, biological assessment data can measure the biological benefits of addressing those effects. That information helps the public understand what is being protected or what could be restored and whether state or tribal Water Quality Standards (WQSs) (i.e., aquatic life protection) are being met. Typically, with improved understanding of what is at stake, the public is more informed, motivated, and engaged in working with resource managers in setting goals for protection or restoration and design solutions that work.

Over the past four decades, state and tribal water quality programs in the United States have used technical tools and information on biological condition to support management decisions. Development of practical methods and technical approaches for biological assessment programs includes field testing by state and tribal programs. These technical advancements build upon existing approaches and can be used to establish or strengthen existing biological assessment and biological criteria programs.

Alignment of examples with the Framework step-by-step approach

The Framework step-by-step approach identifies 9 steps, with 3 having sub-steps. In »Figure 5.16, the examples that clearly discuss a given step (or sub-step) have been identified. A dark gray box indicates a case that is a good example of a Framework step. A light gray box indicates a case that is a good example of a Framework sub-step. For each step, a single case that serves as a good example of that step is identified with a star. A brief description of each step, and the case study element discussing that step are provided after the table. Those cases of this particular example that advocate a need to 'Cycle' back to Step 1 after completion of Step 9 is completed are also highlighted in the line "cycle" of »Figure 5.16.

Figure 5.16: Alignment of Framework steps with case studies from the United States.



Examples of each step in the Framework from the US case studies

Step 1: **Agree vision and set measurable objectives**

Case Study 14 - The Virginia Department of Conservation and Recreation and Virginia Commonwealth University Center for Environmental Studies are collaboratively developing and implementing an online, interactive database that evaluates the ecological integrity of Virginia's streams. The project focusses on the capacity of stakeholders to collaboratively develop water quality goals and implement strategies to achieve them. This case study is an example of extensive community engagement in establishing and achieving WQGs and targets for freshwater ecosystems.

Step 2: **Classify inland water ecosystems**

Case Study 2 - The state of Arizona undertook a quantitative classification of stream biotic communities within a local biogeographical context and subsequently established condition criteria, summarized in lookup tables, for each of the stream categories. This case study is an example of how WQGs for freshwater ecosystems should be established within the context of natural biogeographical variability.

Step 3: **Set basin context**

Case Study 3 - The state of Maryland has developed Geographic Information System (GIS) shapefiles that are available to help local planners locate high-quality waters within their jurisdiction. This case study is an example of the importance of collating and providing existing spatial information to stakeholders in order for them to establish the basin context within which they are working to achieve water quality targets for freshwater ecosystems.

Step 4: **Screen at desktop level for stressors and state of inland water bodies**

Case Study 4 - The Pennsylvania Department of Environmental Protection (PA DEP) calibrated a quantitative Biological Condition Gradient (BCG) using pre-existing data in order to place streams into an appropriate category ranging from reference to heavily stressed conditions. This case study is an example of an initial screening of conditions and stressors throughout a management area to help set benchmarks and guidelines for water quality.

Step 5: **Establish Tier 1- medium resolution or Tier 2- high resolution guidelines for inland water bodies**

Case Study 1 - The state of Maine has established detailed guidelines for multiple biological, habitat, and water quality indicators in order to achieve aquatic life protection goals set out in the Clean Water Act (CWA). A BCG containing four classes of streams is used to set criteria that must be achieved or exceeded for each class. The assessed condition of any water body must not fall below its current class, with the broader goal being to improve the condition of a waterbody to a higher class level. This case study is an example of establishing guidelines for measurable indicators and using these to prevent degradation of water quality and ideally improve it.

Step 6: **Monitor, manage, and synthesize data**

Case Study 17 - The Oregon Department of Environmental Quality and the Oregon Department of Fish and Wildlife have used macroinvertebrate monitoring data to assess the condition of streams throughout Oregon's Coast Coho Evolutionary Significant Unit (ESU). Monitoring was conducted throughout four areas nested within the ESU and the data were used to determine the biological condition of streams based on comparisons between observed monitoring data and those expected based on a multivariate predictive model. This information has since been used to develop a stressor-response model and to guide policy and management decisions. This case study is an example of how monitoring data can be utilized for a number of purposes within a WQGs framework.

Step 7: **Evaluation, category assignment, reporting, and communication**

Case Study 7 - The Iowa Department of Natural Resources determined that a stretch of the North Fork Maquoketa River was not meeting its aquatic life selected targets. They used monitoring data to evaluate the biological condition of the river over several years and identified significant stressors in the watershed. The reports were used to communicate the findings to stakeholders, and the project specifically encouraged residents and businesses in the watershed to take action to improve water quality. This case study is an example of how reporting and communication of monitoring and assessment data to stakeholders is important in implementing an achieving WQGs and targets for freshwater ecosystems.

Step 8: **Integrate guidelines into governance framework and establish process for adaptive management**

Case Study 13 - The state of Ohio uses the CWA within their governance framework to regulate activities associated with discharge of dredge and fill material into waterways. Dredge and fill activities must not violate Ohio WQSs and an index measuring habitat quality is used to set guidelines, assess impacts, and identify required management actions. This case study is an example of incorporating legal frameworks, stakeholder responsibilities, and future management actions into a governance framework to set and achieve WQGs and targets.

Step 9: **Identify cost-benefit considerations and funding issues**

Case Study 8 - The Connecticut Department of Environmental Protection implemented total maximum daily load criteria for Eagleville Brook and as a part of the project investigated the most cost-effective management actions to reduce impervious surface cover, which was identified as a significant stressor on aquatic life of the system. This case study is an example of the importance of considering the cost-effectiveness of management actions given the funding issues often faced by watershed managers.

Example V: A description of the new Indonesian Water Quality Guideline for Ecosystems under development to advance lake management

Introduction and objective setting (Step1)

Water resources in Indonesia are about 6 % of the world water resources or about 21% of total water resources in the Asia Pacific region (WEPA, 2012). There are 5,590 major rivers in Indonesia (WEPA, 2012), and 500 major lakes and thousands of small lakes (MoE, 2008). Most rivers and lakes have experienced pollution due to wastewater discharges and other stressors (WEPA, 2012). A framework for environmental management of water in Indonesia has been stated under an amended law concerning environmental protection and management (Law No.32 of 2009; WEPA 2009). The objective of environmental management in Indonesia is to enable environmentally-sustainable development. Water has been recognized as having an important function in achieving this development and maintaining the well-being of humans and other creatures, and it needs to be wisely managed for the benefit of both present and future generations, as well as to achieve an ecological balance (WEPA, 2012).There is no established WQG for ecosystems, however under the law of water quality management and water pollution control (PP no. 82, 2001), a WQS (Table 5.7) was established for different water use classes aiming to protect water bodies for human use and also the integrity of biota as described in »Table 5.6. The law does not have a framework for specific water body types, which makes it ineffective to protect inland water ecosystems. However, the effort to establish the WQG for a specific ecosystem such as lake was initiated. The Ministry of Environment (MoE) developed a draft of WQG for lake management to evaluate the ecological status of national lake’s ecosystem (MoE, 2010).

Following is a description on a lake management in Indonesia that reflects the Framework approach. One specific example study (Lake Maninjau) will be further discussed.

Facing the threat of lake ecosystem degradation due to pollution, eutrophication, invasive macrophyte coverage and the loss of biodiversity (especially endemic and local/indigenous species) the MoE selected 15 lakes as priority lakes to be restored and protected (MoE, 2010).

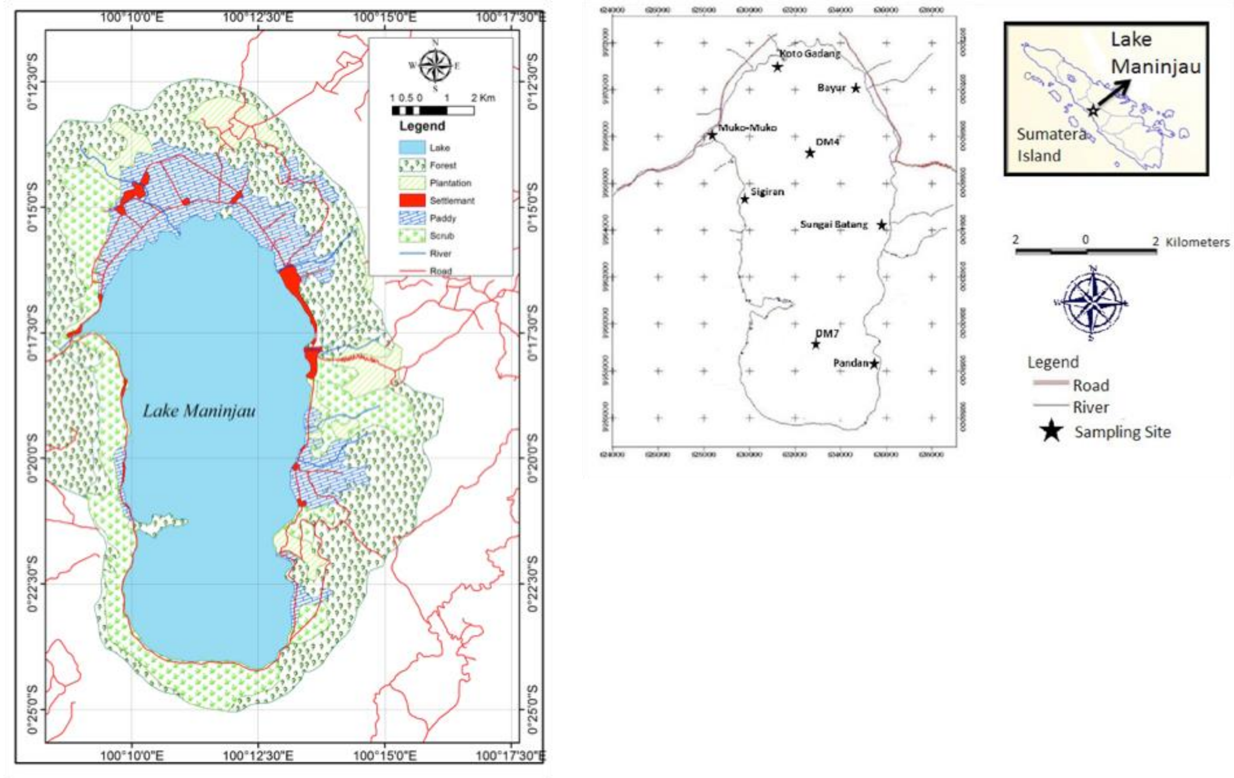
There are two main long-term objectives for lake management; first is to restore degraded lakes to a good/moderate ecosystem health condition where the ecosystem can function ecologically while supporting its Ecosystem Services (ES) and second is to protect and maintain high integrity ecosystem health of lakes that are habitat for endemic or potential indigenous species.

Lake management example case: Lake Maninjau, West Sumatra.

Lake Maninjau is one of the big lakes in Indonesia located in the Western area of Sumatra Island (»Figure 5.17). Lake Maninjau has a surface area of 9737.5 ha and a maximum depth of 165 m. The lake is located at 461.50 m above sea level. Lake Maninjau has several inlets of small streams and hot springs from the mountains. The lake was a national and international tourist destination due to its natural beauty. However, since 2006 due to its deteriorated water quality, only few international tourists have visited the lake.

Figure 5.17 Bathymetry map of Lake Maninjau

Source:provided by the Research Center for Limnology - Indonesian Institute of Sciences (LIPI) in 2015.



Lake Maninjau is a tectono-volcanic type basin, which has been exploited for cage aquaculture since 1992. Hydropower was built in the lake in 1983 and the only water surface outlet was closed and therefore the lake outflow was through the turbine which was basically only deep water layer flowed out of the lake. The lake water flow and level has changed but the residence time only slightly changed (Apip, 2003). The lake is also habitat for endemic and indigenous species. The net cage aquaculture activity has decreased the lake water quality due to excessive nutrients and organic matter inputs from fish pellets and waste. Ecosystem disaster such as massive fish kills occurred in 1997 (Syandri, 2000). Prior to the fish kill incidence, rigorous research on the lake was conducted in 2001 by Research Center for Limnology of Indonesian Institute of Sciences (LIPI), to restore and monitor the lake. The lake surface was covered by toxic filamentous cyanobacteria known as Microcystis trapped in the surface of lake outlet (RCL, 2001). Filamentous cyanobacteria bloom was first observed in 2000 and it could be spotted from the satellite image.

The local government set the objective to restore Lake Maninjau by creating a lake management and monitoring program and planning a new local government regulation to control the cage net aquaculture activity in the lake. Short-term objective was to deal with the eutrophication especially to clean up the lake from Microcystis bloom. This refers to step 1 of the Framework. The long-term objective is to restore the lake from eutrophic to mesotrophic condition by limiting the number of fish cages to reduce the nutrient and organic carbon inputs into the lake and to set a benchmark for WQS for several key indicators. Followings are the vital recommendations made based on the scientific results in 2001:

- Reopen the outlet gate for surface water outflow
- Limit and regulate the number of fish net cages based on the lake carrying capacity to
- Locate the major fish cages near to the outflow area.

The outlet gate (about 1 meter wide) for surface water outflow was built in 2001. The filamentous Microcyctis had been washed out by 2001 (»Figure 5.18 right). Recommendation for the fish cage number allowed was around 7000 units and the placement of major fish cages should be 100 m from the shoreline area and around the area which is close to the outflow area (RCL, 2001). The recommendations would be the ground for planned regulation by the local government.

The condition of the lake indeed improved to mesotrophic condition in 2005 to 2006 (Henny and Schulung, 2012). However, due to the regulation to limit the fish cage number has not been validated and implemented yet, the cage aquaculture activity has been increasing and incontrollable since 2007 up to now. The lake condition has been in low level of ecosystem health again. High concentrations of Chlorophyll a in lake water were observed on almost all surface areas of Lake Maninjau (»Figure 5.19). The lake trophic status changed from oligotrophic-mesotrophic to eutrophic (Henny and Nomosatryo, 2012). Population of filamentous cyanobacteria counted 40 up to 60% in almost all areas especially near the cage net acuaculture (Sulastri et al, 2012). Filamentous cyanobacteria bloom was observed again in 2013 and it was also spotted from the satellite image (»Figure 5.18).

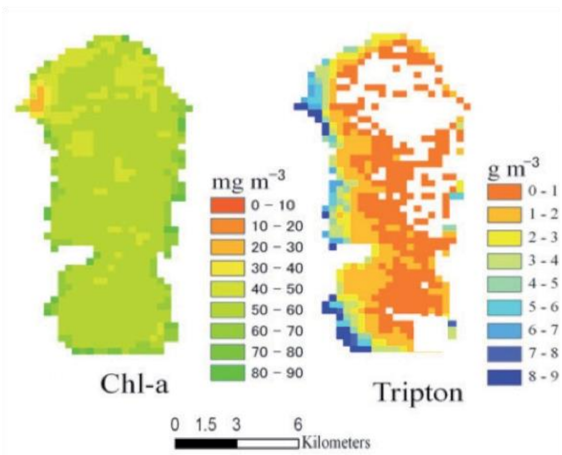
Figure 5.18: Satellite images of algal blooming in Lake Maninjau in the years 2000, 2001 and 2013.

Citra LANDSAT 5/TM. Date 2000 (algae blooming-left); Citra LANDSAT7 ETM+ Aq. Date 2001 (middle); Citra LANDSAT-8 Aq. Date May 17th, 2013 (algae blooming-right).

Source: US Geological Survey (USGS, 2013).

Figure 5.19: Distribution map of Chlorophyll a and tripton concentrations in surface water of Lake Maninjau.

Source: Setiawan et al. (2012).



Classifications of the lake's ecosystem (Step 2)

Since there is no official WQG for ecosystem yet, an assessment for inland water quality in Indonesia is usually for water use purpose only not for ecosystem health per se. Indonesia has only a WQS to conserve its water bodies for its water use (government regulation PP No. 82, 2001). Class of Water Quality Criteria (WQC) for inland water use (»Table 5.6) were established in WQS according to the benchmark of selected physical chemical indicators. These indicators will be discussed further in the »Section 5.6.4.

Table 5.6: Classification of water use based on a water body quality according to the regulated physical chemical Water Quality Standard (PP no.82, 2001).

Source: MoE (2011).

Class	Use
Class I	Water can be used as raw water for drinking water, and/or other usage that requires the same water quality for such usage
Class II	Water can be used as raw water for recreation, fishery, animal husbandry, irrigation, and/or usage that requires the same water quality for such usage
Class III	Water that can be used for fishery, animal husbandry, irrigation, industry and/or other usage that requires the same water quality for such usage
Class IV	Water that can be used for irrigation, industry and/or other usage that requires the same water quality for such usage

As for lake assessment, a draft of lake management guideline made by MoE (2010), although it is not officially published yet, has been used to classify the ecosystem health of lakes. The lake ecosystem in Indonesia includes natural and man-made lakes (i.e. small, big lakes and reservoirs). Under its lake management program, MoE aimed to restore and to protect the ecosystems of fifteen "priority" large lakes (2013). The typology of the fifteen lakes is based on their origin and includes tectonic, tectono-volcanic, volcanic, floodplain, oxbow and solution (karst depressions) lakes. Lake Maninjau for instance is a caldera lake but the lake is also affected by tectonic activity and therefore it is considered as a tectono-volcanic type of lake[64]. The ecosystem health of each priority lake selected would be classified according to its typology.

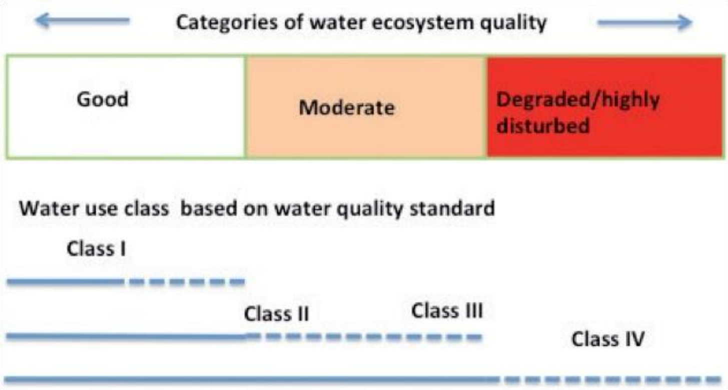
In the draft of lake management guideline, a classification of lake's ecosystem health was set based on selected indicators of lake's hydromorphology, lake water physics, chemistry and biology. Based on the benchmark of the indicators, the lake ecosystem health has been classified into three categories i.e. 1. Good, 2. Moderate/disturbed and 3. Degraded ecosystems (»Figure 5.20). The indicators are explained in the »Section 5.6.4. As for physical and chemical parameters, the benchmark used to set the ecosystem health class followed the WQS classification (»Table 5.6 and »Table 5.8).

For example based on the assessment of the trophic status of lake's ecosystem, the condition of oligotrophic is in the category of good ecosystem health, mesotrophic condition is in the category of moderate or slightly disturbed ecosystem and eutrophic–hypereutrophic condition is in the category of highly disturbed ecosystem/impaired ecosystem. Based on the categories of water quality, good water quality ecosystem supports class I water use, moderate ecosystem quality can support class II and class III water use and degraded ecosystem quality only supports class IV water use.

[64] http://danau.limnologi.lipi.go.id

Figure 5.20: Different classes of water utilization along different freshwater ecosystem quality status in Indonesia

Source: provided by Research Center for Limnology - Indonesian Institute of Sciences (LIPI) in 2015.



Desktop screening (Step 4)

The 15 priority lakes were selected based on physical-chemical water quality and also biological data such as fish and phytoplankton (MoE, 2011). For most lakes the major pressures and stressors that cause lake degradation have been identified. However, the database available comprises only certain indicators and short-term time series. Lake Maninjau has been the most studied and monitored, however data are still insufficient to develop a lake water quality model .

Screening of pressures and stressors (Step 4a)

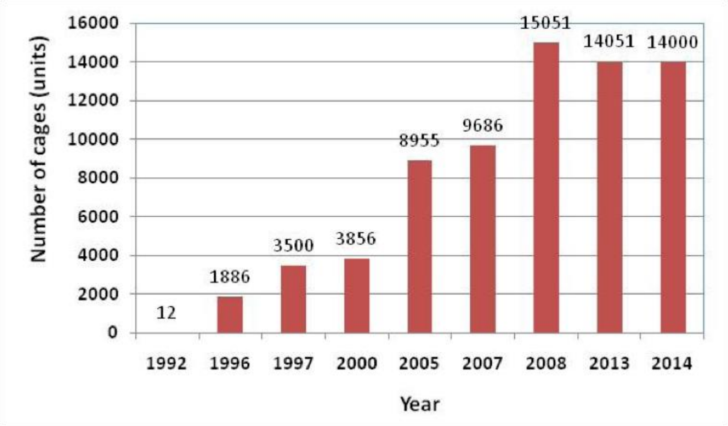
Increasing population and human needs have put multiple pressures on the freshwater ecosystems of the fifteen priority lakes. Most of large lakes in Indonesia have been dammed for hydropower generation. Land use/cover change in the watersheds developed for agricultural area and human settlement have disturbed the lake ecosystems structurally and functionally. One of the major pressures on the lake ecosystems is net cage aquaculture in which excess fish pellet residues and feces have been the major inputs of nutrients and organic matter (organic carbon) to the lake (Sulastri et al, 2012). It becomes evident that the main problem in Lake Maninjau is excessive amount of nutrient and organic carbon which stimulate eutrophication and microbial activity, which increases the production of toxic gases such as hydrogen sulphides and ammonia (Henny and Nomosatryo, 2012). The database of Lake Maninjau is available via the information system of Indonesian lakes (SIDI) including information on hydromorphology, land use/cover on catchment area, utilization, biodiversity and the water quality.

Growing numbers of fish cages exceeding the lake carrying capacity in Lake Maninjau since 1990s has lead to ecosystem disaster such as deteriorated lake water quality and mass and fish kills problem. The number of fish cages has increased from less than 100 units in 1992 to more than 10,000 units in 2008 up to recent time (»Figure 5.21). Fish kill incidences increased after 2008, with multiple incidences per year. High number of fish cages is believed to be the cause of these incidences. Since 2005 the number of cages has exceeded the numbers that were allowed based on the previous studies by RCL (2001) as results of no enforced regulation on that yet.

High input of organic waste of excessive fish pellets from cage aquaculture over the years has caused the accumulation of organic matter in the lake sediment. Concentration of organic content as Chemical Oxygen Demand (COD) in the water increased around 80% from 2006 to 2009. Total and volatile solids in the sediment of Lake Maninjau increased from 2006 to 2009 by more than 70%. Organic carbon concentration in the lake sediment reached 700 g/kg (Henny, 2009). Since the lake contains sulfur, the excess organic carbon stimulates microbial activity resulting in elevated hydrogen sulphide concentration, causing not only oxygen depletion but also internal phosphate release from the sediment into the water column. Lake Maninjau with a maximum depth of 168 m has now the oxycline layer between 10 to 15m of depth indicating that anoxic hypolimnion is expanding to the upper water column close to the surface (Henny, 2014).

Figure 5.21: Number of fish cages in 12-year periods for a Lake Maninjau.

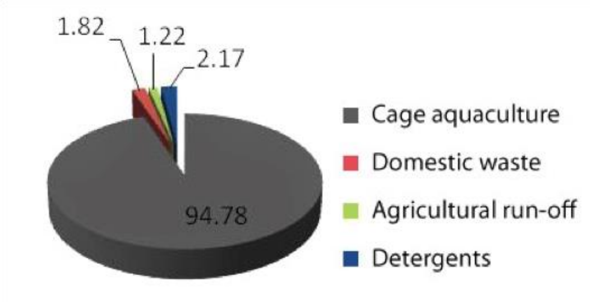
Source: Henny (2014).



The cage aquaculture activity also has been causing a large amount of nutrient inputs in the Lake Maninjau. Nutrient input in the lake from cage aquaculture was estimated around 400 tons/year in 2006. Ninety five percent of the nutrient waste load, i.e. nitrogen and phosphorus that enters the lake were from cage aquaculture, while only a small percentage of wastes from other sources such as domestic waste, agricultural run-off or detergents (»Figure 5.22).

Figure 5.22: Total and percentage of nitrogen and phosphorus waste load to Lake Maninjau in 2006 stemming from cage aquaculture, domestic waste, agricultural run-off and detergents.

Source: http://danau.limnologi.lipi.go.id



The status of lake water quality could not be assessed based on certain physico-chemical indicators only. As mentioned earlier, a draft of the WQG for lake management has included more key indicators for the lake assessment, e.g. hydromorphology, biological indicators, and even the trophic status, and other key indicators to classify lake´s ecosystem health (MoE, 2013). The indicators concerned are as follows:

- Hydromorphology
- Trophic status (oligotrophic, mesotrophic and eutrophic/hypereutrophic)
- Water quality (physical and chemical parameters)
- Biodiversity (flora and fauna – endemic/local species, invasive species)
- Food web (balanced trophic level – primary, secondary productivity, and consumers)

- Eutrophication (percentage of macrophyte coverage; blue green algae/microcystis)

- Carrying capacity (based on phosphor and DO concentrations in lake water)

The criteria or the benchmark values for each indicator were set according to good, moderate and highly disturbed ecosystem quality status categories. However, the benchmark for physical and chemical indicators follows the regulated WQS (MoE, 2001). Based on the lake assessment results of the 15 priority lakes one lake is in good ecosystem status, nine lakes are in moderate ecosystem status but at high risk condition to become disturbed and five lakes are in highly disturbed ecosystem status (MoE, 2012). Lake Maninjau has been classified as a highly disturbed ecosystem.

Beside the lake management guideline, the MoE was also proposing the revision of existing WQS to include several important indicators such as water transparency (Secchi depth) and Chl-a (MoE, 2011). Both, the guideline draft as well as its revision have never been officially published. For Lake Maninjau, which faces problems caused by the pressures in the catchment (land use cover change) and by stressors in the lake by cage aquaculture a Tier 2 approach from the Framework can be used for the assessment of the ecosystem quality status. The lake faces eutrophication problems and experienced frequent fish kills. The objective of the lake management program is to restore the water quality to a mesotrophic condition, which is in a “good ecosystem health” category with water quality (based on physical chemical indicators) for water use is to be in class II. The key indicators for the level of eutrophication are TN, TP, Chl-a, and Secchi depth. In fact a benchmark of these parameters had been set for Lake Maninjau based on 2001-2002 data obtained by RCL (2002). Other important indicators considered to be included are sulphides total and DO to reduce fish kill incidences. Population of endemic and local indigenous fish species are biological indicators to be included in lake assessment (RCL, 2014).

From monitoring to policy making (Steps 6-9)

Up to now, monitoring of water quality of water bodies in Indonesia in general is based on physical and chemical parameters only. Monitoring is usually implemented subsequent to pollution events and the indicators selected are based on the nature of the pollution source. The monitoring data obtained are then compared with the benchmarks of the WQS (government regulation PP no. 82, 2001) and the water body is assigned according to the water use class. When the water quality of a water body is in Class I or II the water body is perceived as in good condition. Most water bodies face increasing problems and most of the water bodies are in the “highly disturbed condition” (Class IV) and are ecologically not functional (MoE, 2012).

For Lake Maninjau, monitoring has been implemented at least three times a year including monitoring of physical and chemical parameters (i.e. pH, temperature, conductivity), the key indicators including TP, Chl-a, Secchi depth, sulphides total and DO and sometimes population of endemic and local indigenous fish species. However, due to data gaps water quality modelling has not been developed yet. In 2001 the assessment of the lake’s carrying capacity based on TP loads from cage aquaculture and lake water concentration of TP was done in order to estimate the number of cages that can be authorized without profound impact on the lake ecosystem. For the example of Lake Maninjau, the number of cages allowed is at around 7000 fish cages (RCL, 2001). However since the regulation planned to limit the fish cage number has not been validated yet, the cage numbers are still high continuously persevering lake eutrophication problems including fish kill incidences. In summary, the restoration program for Lake Maninjau has not been fully implemented and successful so far, although the monitoring program is still being performed.

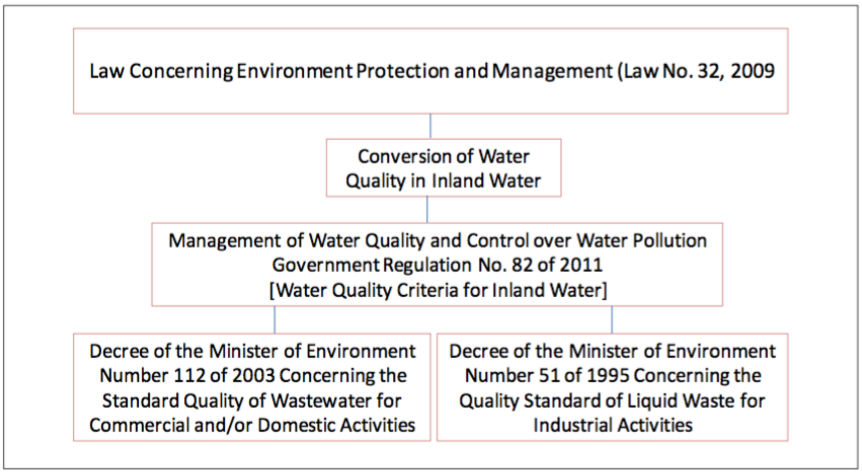
Inland water ecosystems are to be considered as water resources is regulated under the Law No.32 of 2009 (»Figure 5.23). The only existing regulation that has been used as a legal basis for inland water ecosystem assessment is a government regulation for water quality management and water pollution control (PP no. 82, 2001).

Towards national guidelines and legistation for freshwater ecosystem health

»Figure 5.23 gives an overview on the general legal basis of water environment management in Indonesia, which covers the laws and regulations concerning freshwater ecosystem quality management. Management of water quality in Indonesia is divided within three levels of governmental structures: Central (national/trans-boundary level), Provincial (provincial level) and District/City (district/city level). These include carrying out water quality monitoring for the inland water ecosystem. Budget insufficiencies, capacity development needs and weak implementation of existing regulations remain issues that challenge the respective administrative level and might lead to so far unsuccessful lake management and restoration programs

Figure 5.23: Legal basis of water environment management.

Source: WEPA (2012).



In the case of Lake Maninjau, the local government has urged the regulation to limit the number of fish cages allowed according to lake's carrying capacity. Although a decree by local district government (Penerbitam Perbub No. 22, 2009) for Lake Maninjau management and monitoring program has been published in order to restore the lake, its implementation has not yet been imposed (RCL, 2014).

Under new government of Republic Indonesia, the MoE has been merged with Ministry of Forestry in 2015. The inland water quality monitoring and evaluation program will be under new administration of MoE. The MoE has taken seriously to restore and improve the ecosystem quality of inland water. MoF will evaluate the existing WQS and develop a guideline for ecosystem water quality for each type of inland water ecosystems along with a monitoring program and the implementation of restoration technology to maintain inlnd water ecosystems health. A framework to protect inland water ecosystems is needed and therefore a WQG for ecosystems of inland waters (rivers, lakes and groundwater) is being prepared along with a draft of ministerial decree (MoE) to control the impairment of inland water ecosystems (MoE, 2015). As for the lake management and monitoring program, WQC for each key indicator including physical, chemical and biological indicators will be established according to each lake typology and its preset objective. Indonesia can therefore apply step by step approach of the Framework to help develop a WQG to maintain good ecosystem health quality for its inland water ecosystems.

Table 5.7: Indonesian water quality standards as regulated in government regulation PP No. 82 of the year 2001 (MoE, 2001) and plans for revisions or additions to that 2001-regulation by the Ministry of Environment in 2011 (MoE, 2011). The revisions and adaptions of the 2001 standard are not legally binding yet; however, they document that Indonesia is getting serious to amend its water quality standards not only for water uses but for ecosystems as well.

Parameter	**Unit**	**Class**				**Revision of WQ parameter**
		I	**II**	**III**	**IV**	
PHYSICAL						
Temperature	oC	Deviation ±3 of the	Deviation ±3 of the	Deviation ±3 of the	Deviation ±3 of the	PP no. 82, 2001
TDS	mg/L	1000	1000	1000	2000	PP no. 82, 2001
TSS	mg/L	50	50	400	400	PP no. 82, 2001
Light regime (secchi depth)	m	10	4	2.5	<2.5	Additional for lakes and
Colour	Pt/Co	15	50	100	150	Additional (not applied for
CHEMICAL						
pH	-	6–9	6–9	6–9	5–9	PP no. 82, 2001
BOD	mg/L	2	3	6	12	PP no. 82, 2001
COD	mg/L	10	25	50	100	
	mg/L	10	25	40	50	Plan for Revision
DO	mg/L	6	4	3	0	PP no. 82, 2001
	mg/L	6	4	3	1	Plan for Revision
Total Phosphate (as P)	mg/L	0.2	0.2	1	5	PP no. 82, 2001
	mg/L	0.01	0.03	0.1	>0.1	Additional for lakes
PO_4^{-} -P	mg/L	0.2	0.2	1	5	PP no. 82, 2001
Total N	mg/L	0.65	0.75	1.90	> 1.90	PP no. 82, 2001
NO_3-N	mg/L	10	10	20	20	PP no. 82, 2001
NH_3-N	mg/L	0.5	-	-	-	PP no. 82, 2001
	mg/L	0.1	0.5	1	1.5	Plan for Revision
NO_2^{-}-N (nitrite)	mg/L	0.06	0.06	0.06	0.06	PP no. 82, 2001
Chloride	mg/L	600	-	-	-	PP no. 82, 2001
	mg/L	300	300	300	600	Plan for revision
Cyanide (CN)	mg/L	0,02	0,02	0,02	-	PP no. 82, 2001
Fluoride	mg/L	0,5	1,5	1,5	-	PP no. 82, 2001
Sulphate	mg/L	400	-	-	-	PP no. 82, 2001
	mg/L	300	300	300	400	Plan for Revision
Free Chlorine	mg/L	0.03	0.03	0.03	-	PP no. 82, 2001
H_2S	mg/L	0.002	0.002	0.002	-	PP no. 82, 2001
Na	%	60	60	60	60	Additional
SAR	-	18	18	18	18	Additional
Chl-*a*	µg/L	10	50	100	200	Additional for lake and
METALS AND HEAVY METALS						
Arsen (As)	mg/L	1	1	1	-	PP no. 82, 2001
	mg/L	0.05	0.05	0.05	1	Plan for Revision
Cobalt (Co)	mg/L	0.2	0.2	0.2	0.2	PP no. 82, 2001
Barium (Ba)	mg/L	1	-	-	-	PP no. 82, 2001
Boron (Bo)	mg/L	1	1	1	1	PP no. 82, 2001
Selenium (Se)	mg/L	0.01	0.05	0.05	0.05	PP no. 82, 2001
Cadmium (Cd)	mg/L	0.01	0.01	0.01	0.01	PP no. 82, 2001
Chromium (VI)	mg/L	0.05	0.05	0.05	0.01	PP no. 82, 2001
	mg/L	0.05	0.05	0.05	0.05	Plan for Revision
CoGRer (Cu)	mg/L	0.02	0.02	0.02	0.2	PP no. 82, 2001
Iron (Fe)	mg/L	0.3	-	-	-	PP no. 82, 2001
	mg/L	0.3	0.5	1	1.5	Plan for Revision

Lead (Pb)	mg/L	0.03	0.03	0.03	1	PP no. 82, 2001
	mg/L	0.03	0.03	0.03	0.05	Plan for Revision
Mangan (Mn)	mg/L	0.1	-	-	-	PP no. 82, 2001
Mercury (Hg)	mg/L	0.001	0.002	0.002	0.002	PP no. 82, 2001
Zinc (Zn)	mg/L	0.05	0.05	0.05	2	PP no. 82, 2001
PATHOGENS						
Fecal Coliform	Jml/100	100	1000	2000	2000	PP no. 82, 2001
Total Coliform	Jml/100	1000	5000	10000	10000	PP no. 82, 2001
RADIOACTIVE						
Gross A	Bq/L	0,1	0,1	0,1	0,1	PP no. 82, 2001
Gross B	Bq/L	1	1	1	1	PP no. 82, 2001
ORGANIC POLUTANTS						
Oil and Grease	µg/L	1000	1000	1000	-	PP no. 82, 2001
Detergen as MBAS	µg/L	200	200	200	-	PP no. 82, 2001
Phenol	µg/L	1	1	1	1	PP no. 82, 2001
BHC	µg/L	210	210	210	-	PP no. 82, 2001
Aldrin/Dieldrin	µg/L	17	-	-	-	PP no. 82, 2001
Chlordane	µg/L	3	-	-	-	PP no. 82, 2001
DDT	µg/L	2	2	2	2	PP no. 82, 2001
Heptachlor and Heptachlor Epoxide	µg/L	18	-	-	-	PP no. 82, 2001
Lindane	µg/L	56	-	-	-	PP no. 82, 2001
Mothoxychlor	µg/L	35	-	-	-	PP no. 82, 2001
Endrin	µg/L	1	4	4	-	PP no. 82, 2001
Toxaphan	µg/L	5	-	-	-	PP no. 82, 2001

Example VI - Application of South African Water Quality Guidelines for aquatic ecosystems in the context of the Reserve and Resource Quality Objectives

The context - South African Water Act and water resource management

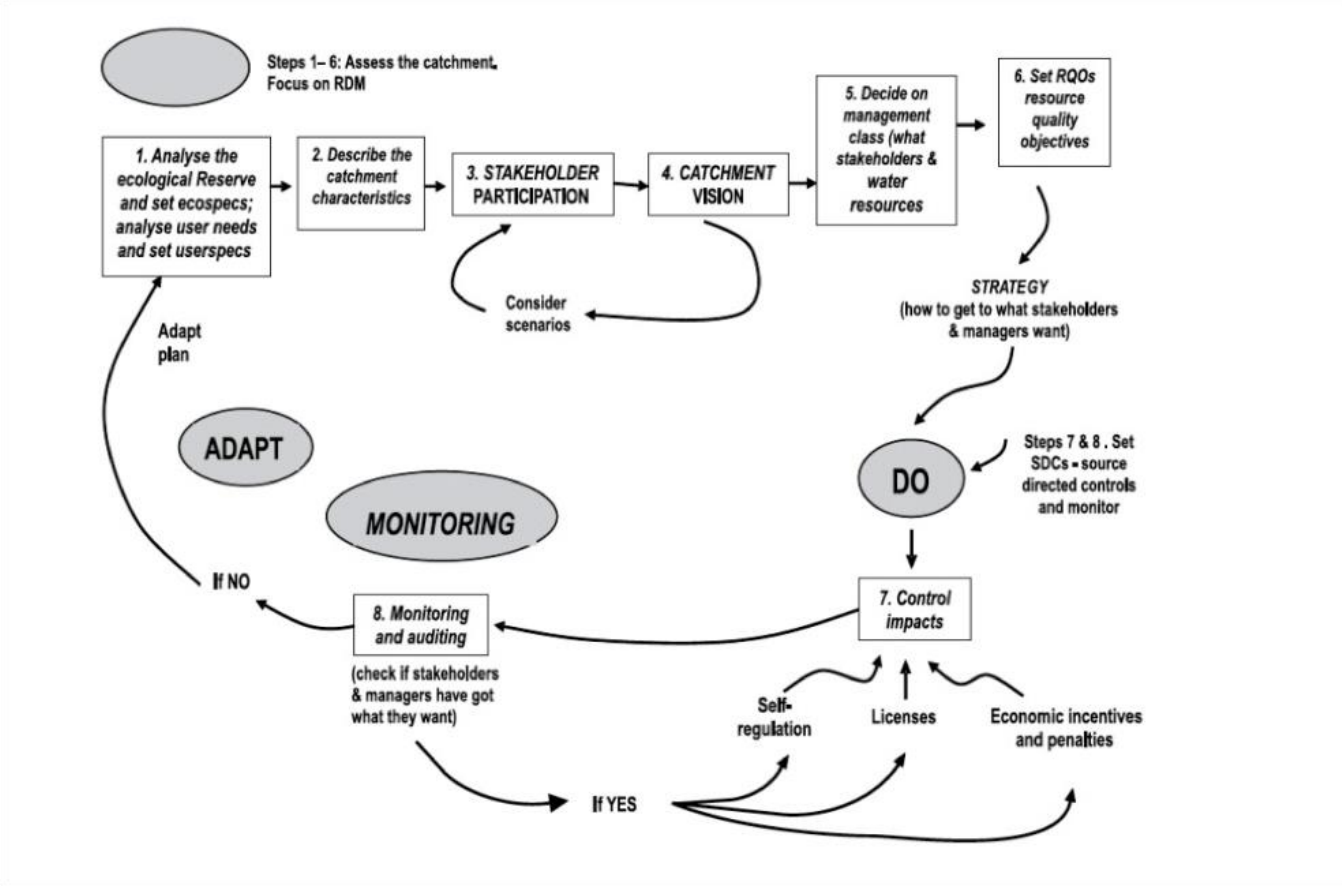
In South Africa, a semi-arid and water scarce country, the leader and regulator of the water sector is the Department of Water Affairs (DWA), with the mandated responsibility "to ensure that all people have access to sustainable water services and resources" (p. iii, DWA Directorate of Water Services Planning and Information 2013, which provides a strategic overview of the water sector). The DWAF is governed by the South African Water Services Act of 1997 and the National Water Act (Act No. 36 of 1998, NWA). It shares river health and related catchment management functions with the Department of Environmental Affairs. The NWA aims to ensure that all water resources are used, managed and controlled in such a way that they benefit all users. It prescribes a series of measures, to be developed progressively within the context of the National Water Resource Strategy and individual catchment management strategies, intended to ensure comprehensive protection of water resources so that they can be used in a sustainable manner.

In total, as defined under the NWA, the Reserve comprises the amount of water required to sustain the ecosystem (the Ecological Reserve) and the amount of water needed to provide for basic human needs (the basic Humans Need Reserve). The Reserve is the only water right specified as inviolable in the law, with water for basic human needs having the highest allocation priority (DWA 2011). Ecosystems are central to the NWA (Palmer et al. 2004a) and the main tool for water resource protection is the Ecological Reserve (i.e. the environmental water allocated; see »Box 2.2 Environmental flows) which is defined in the Act as: "the quantity and quality of water required to protect ecosystems in order to secure ecologically sustainable development and use of the relevant resource" (chapter 1(10(xviii)(b)) in NWA 1998).

During the assessment of the Ecological Reserve, based on stakeholder objectives for the water body of concern, an interdisciplinary team of scientists use a structured process to derive quantitative and qualitative instream objectives for flow, water quality variables, habitat and biota. The water quantity aspects of Ecological Reserve determination are outlined in Palmer 1999 and several of the desktop planning and holistic methodologies commonly employed are outlined in Tharme (2003), and detailed in Hughes and Hannart (2003), King et al. 2000 (Building Block Methodology, which includes a specific chapter on water quality), and Brown et al. 2013 (Downstream Response to Imposed Flow Transformations). For further reading on the integration of water quality, Palmer et al. (2005) describe the development in South Africa of water quality methods within ecological Reserve assessments and the links to environmental flows. The interrelationships among ecotoxicology, biomonitoring, and water chemistry in the integration of water quality into environmental flow assessments are discussed by Scherman et al. (2003). Hughes (2004) describes an integrating framework for Ecological Reserve determination and implementation, incorporating water quality and quantity components for South African rivers.

Three approaches constitute the basis of the national plan to implement sustainability through resource protection: (1) water resource classification; (2) Resource-Directed Measures (RDM), which includes quantification of the ecological Reserve; and (3) Source-Directed Controls (SDC) (see »Figure 5.24).

Figure 5.24: A proposed strategy for undertaking IWRM from a water quality perspective. Water resource classification (steps 1 and 5) is an integral part of Resource-Directed Measures (RDMs) which together with Source-Directed Controls (SDCs) combine to achieve IWRM.
Source: Palmer et al. (2004).



Classification of significant water resources

Water resource classification is the process of classifying the continuum of 'health' or integrity of the resource, from natural state, equated with excellent condition, through to degraded or poor condition (Palmer et al. 2004a). Each significant water resource (surface water, wetlands, groundwater and estuaries) is classified according to a national water resource classification system, at the sub-catchment scale of units (e.g. a river reach between a dam and a major tributary downstream of which water quality and/or ecological health might change) (see »Figure 5.25 below). The Reserve determination contributes information to the classification of the resource. This classification, which is an integral part of the RDM process (see DWAF RDM Manual, DWAF 2003) results in a management class and associated of Resource Quality Objectives (RQOs) for each water resource, which gives direction for future management activities in each Water Management Area (WMA). South Africa has nine WMAs responsible for water management in their areas of the country, of which two have already been decentralized to Catchment Management Agencies (CMAs) (DWA, 2013).

Figure 5.25: Diagram of a proposed system of water resource classification. Each ecological category (A-E or Excellent-Poor) is defined by numerical and descriptive objectives termed ecospecs. These are combined with the requirements of users (termed userspecs) into Resource Quality Objectives (RQOs), which define a set of associated management classes. Generally, the use of the A-E classification is restricted to defining environmental categories, whereas the Excellent-Poor nomenclature has been used to define water quality ecospecs as well as to describe management classes that combine both userspecs and ecospecs. The classification system is still being refined.

Source: Palmer et al. (2004).

EXCELLENT	GOOD		FAIR	POOR	MANAGEMENT CLASSES/
A Minimal	B Slight	C Moderate	D Heavy	E & F Unacceptable	Ecological Categories User impact
Unmodified	Slightly modified	Moderately modified	Considerably modified	Critically modified	Ecological condition

The purpose of the RQOs is to establish clear goals relating to the quality of the relevant water resources, with balance sought between the need to protect and sustain water resources and the need to use them (DWA, 2011). Thus, the active part of the water resources classification is the RQOs produced. Setting RQOs for a chosen management unit of a water resource, is a technical process of integration of water quality, water quantity and ecosystem integrity (as further discussed in DWA, 2011). The RQOs are the numerical and narrative descriptors of conditions that need to be met in order to achieve the required management scenario as provided during resource classification. Such descriptors relate to the:

c) quantity, pattern, timing, water level and assurance of instream flow

d) water quality including the physical, chemical, and biological characteristics of the water

e) character and condition of instream and riparian habitat

f) characteristics, condition and distribution of aquatic biota

Resource-directed measures

As illustrated in »Figure 5.24, and detailed in Palmer et al. (2004a), RDMs comprise a sequence of activities by Government, technical experts and stakeholders, focused on the water resource, from setting of the Reserve through to monitoring and adaptive management. These steps have much in common with many of the the steps outlined in the Framework.

The first step, and a central RDM activity, is to determine the Reserve and to set the numerical and/or descriptive ecospecs for each Resource Unit (RU), in each ecological category, for flow and water quality variables. These ecospec values are also called 'boundary values', because they indicate a change of ecological condition. The water quality ecospecs are equivalent to the in-stream WQGs. The suite of variables typically considered within the RDM process for setting the ecological Reserve includes: system variables (e.g. pH, temperature, DO, TSS); inorganic salts (e.g. sodium chloride, magnesium sulphate); nutrients (e.g. soluble reactive phosphorus, total inorganic nitrogen); and toxic substances (See »Chapter 3 and »Annex 2 list of toxics in SA WQGs). The development of ecospecs for salts (see »Box 5.4) provides an example of one of the challenges in establishing ecologically meaningful tolerance levels for aquatic biota.

Notably, the Department of Water and Sanitation now has new systemic RQOs based on the latest research in this field (DWA, 2011), but the process of calculating the guidelines remains to be published (Prof. CG Palmer, Unilever Centre for Environmental Water Quality, Institute for Water Research, Rhodes University, Grahamstown, South Africa pers. comm. October 2015).

In South Africa, ecotoxicology tools are proving extremely useful for establishing the ranges of tolerance by the biota for various water quality parameters and the associated thresholds of concern. For example, Palmer et al. (2004a) discuss and present a toxicity database derived for freshwater macroinvertebrates that has rapidly evolved during its specific applications towards the protection of South African water resources. Prior to the early 1990s, there were few data on the tolerance ranges of local freshwater organisms to different physical and chemical variables. By 2004, however, a still-growing database was developed in which the responses of some 21 taxa to 26 single-substance pollutants or mixtures are recorded (Palmer et al. 2004a). Ecotoxicity tests have been applied to complex industrial wastes, and to both surface waters and groundwaters.

Salinity is one example where there continues to be rapid advancement in the establishment of thresholds of potential concern in terms of the toxicity of individual salts and specific combinations of salts in South African water bodies. Table 1 of Palmer et al. (2004a) summarizes in detail the results for several toxicants (salts) and species tested, using standard protocols (see Palmer et al. 2004a, for details of the various methods tested) in various river systems across South Africa. The Olifants River system was one of the test case systems; see further below for some details on the establishment of RQOs for this river system.

In the Olifants, an alternative hazard-based method of relating ecotoxicity endpoints to categories of present condition was trialed, after the standard chemical guideline for salinity within the 1996 WQGs of 'proportional changes from local background conditions, with 15% deviation from natural as the management objective' was found to be inadequate when compared with the results from South African Scoring System invertebrate biomonitoring. The toxicity of various salts, including sodium sulphate (a dominant ion in many of the Olifants River reaches) was assessed using the mayfly, Trichorythus discolor, an abundant invertebrate species native to the region's rivers.

Box 5.4: Use of a toxicity database for freshwater organisms in the protection of South African water resources

The user needs or userspecs, when combined with the ecospecs, constitute the RQOs, which define the management classes. When a water user has a more stringent water quality or flow requirement than the ecosystem, and that will not impair it, then the userspec becomes the RQO. After the Ecological Reserve, ecospecs, and userspecs have been determined, the next steps are to describe the catchment characteristics and engage with the stakeholder participation process (»Figure 5.24). With stakeholders, a catchment vision, and management class and appropriate RQOs for each water RU are then set.

Source-directed control measures

Once the corresponding implementation strategy has been developed, subsequent steps represent the management actions needed to ensure that the objectives set in the RDM process are met - these are the SDCs. For water quality these SDCs may include, for example, waste discharge licenses for chemical pollutants or various economic instruments such as penalties and incentives.

An example - Establishing preliminary Resource Quality Objectives for the Olifants WMA

The Olifants WMA includes the Olifants River catchment within South Africa (excluding the Letaba Catchment), a tributary of the Limpopo River. The Olifants WMA falls within the provinces of Gauteng, Mpumalanga and Limpopo, and covers approximately 54 550 km2. Within the Water Resource Classification study the Olifants WMA was divided into 12 Integrated Units of Analyses (IUAs) (based on socio-economic requirements) and 121 RUs were delineated (DWA, 2014). The Olifants River is one of the major water resources in the area and originates in the Mpumalanga Highveld, initially flowing northwards before curving in an easterly direction through the Kruger National Park (KNP) and into Mozambique, where it joins the Limpopo River which discharges into the Indian

Ocean. The catchment includes several mining and industrial centres, some of the largest thermal power stations in the world, areas of extensive irrigation and rain-fed cultivation. It also includes several largely undeveloped areas with scattered rural settlements and traverses a major protected area, the KNP.

In the illustrative case of the Olifants WMA the process was incremental, with the Reserve established in 2004, the classification of significant water resources completed in 2012, and the determination of RQOs undertaken in 2014 (DWS, 2014, DWA, 2014) and ongoing. A summary of the key water quality issues, drivers and effects in the Olifants WMA is given in Table 5.8 below.

Table 5.8: Summary of the main water quality issues identified within the Olifants Water Management Area. Source: DWS (2014) and DWA (2014).

WMA	Water Quality Issue	Driver	Effect
WMA 4: Olifants	Eutrophication (Nutrient enrichment)	Wastewater treatment works, intensive agriculture fertilizer use and dense urban sprawl un-serviced sewage	Algal growth, smell, toxic algae, increased water treatment costs, taste and odour problems, increased irrigation clogging, impacts on aesthetics and recreational water users
	Microbial contamination	Wastewater treatment works and informal dense settlements	Impact on recreational users (human health), washing and bathing
	Turbidity	Informal dense settlements, urbanization, mining, agriculture and point source discharges	Dam sedimentation, increased water treatment costs and irrigation clogging
	Salinisation	Mines (operational and abandoned), wastewater treatment works and agricultural (intense irrigation)	Increased water treatment costs, soil salinity and irrigation system clogging
	Toxicants	Pesticides (subtropical fruits, nuts) industry	Fish kills, bioaccumulation of pollutants in fish and crocodiles and crocodile deaths
	Altered flow regime	Dams and weirs	Turbidity (erosion), algal growth, water temperature increase, dissolved oxygen changes, taste and odour changes, impact on recreational water users, fish kills and changes in environmental flows
	Acid mine drainage	Mines (operational and abandoned) and controlled releases	Mobilisation of metals, fish and crocodile deaths, bioaccumulation of pollutants in fish and crocodiles
	Metal contamination	Mines (operational and abandoned)	Mobilisation of metals, fish kills, bioaccumulation and crocodile deaths in Loskop dam

Procedurally, the RQOs determination for the Olifants WMA involved the application of the seven step framework established by DWA in 2011 (see also »Figure 5.26); this procedure shows a high degree of similarity with a number of the Framework steps described in »Chapter 4. Some of these steps were achieved in the Water Resource Classification study for the Olifants. These steps, used to determine RQOs for the four main water body types assessed in the WMA, namely: rivers, wetlands, groundwater, and dams were (DWS, 2014; DWA, 2014):

Step 1. Delineate the IUAs and RUs.
Step 2. Establish a vision for the catchment and key elements for the IUAs.
Step 3. Prioritise and select RUs and ecosystems for RQO determination.
Step 4. Prioritise sub-components for RQO determination, select indicators for monitoring and propose the direction of change.
Step 5. Develop draft RQOs and Numerical Limits (NLs).
Step 6. Agree RUs, RQOs and NLs with stakeholders.
Step 7. Finalise and Gazette the RQOs.

The components and sub-components for which RQOs and NLs were provided were:

- Quality components including low and high flow sub-components
- Quantity components including nutrients, salts, system variables, toxicants and pathogen subcomponents.
- Habitat components including instream and riparian habitat sub-components.
- Biota components including fish, plants, mammals, birds, amphibians and reptiles, periphyton,
- invertebrates, and diatom sub-components.

A total of 494 RQOs of varying complexity were determined for the Olifants WMA: 212 RQOs for river resources; 80 RQOs for wetlands resources; 69 RQOs were determined for dam resources; and 133 RQOs for groundwater resources. The details for all RUs are provided in DWS 2014, and one result is shown below for wetlands, for illustrative purposes (»Table 5.9).

Table 5.9: Example of a simple Resource Quality Objective (RQO) for wetland water quality in one of the priority Resource Units (RUs) in the Olifants Water Management Area (WMA). IUA = Integrated Unit of Analysis. Source: DWS (2014).

Wetland Water Quality							
IUA	**Wetland**	**RU**	**Component**	**Sub Component**	**RQO**	**Indicator/ measure**	**Numerical Limits**
2	2.3 Dolmas wetland	RU 21	Quality	Pathogen	E. Coli levels must comply with fitness for use guidelines	E. Coli	≤ 130 counts/ 100 ml

Figure 5.26: Summary of the procedure used to determine RQOs for the Olifants River WMA, South Africa.
Source: DWS (2014).

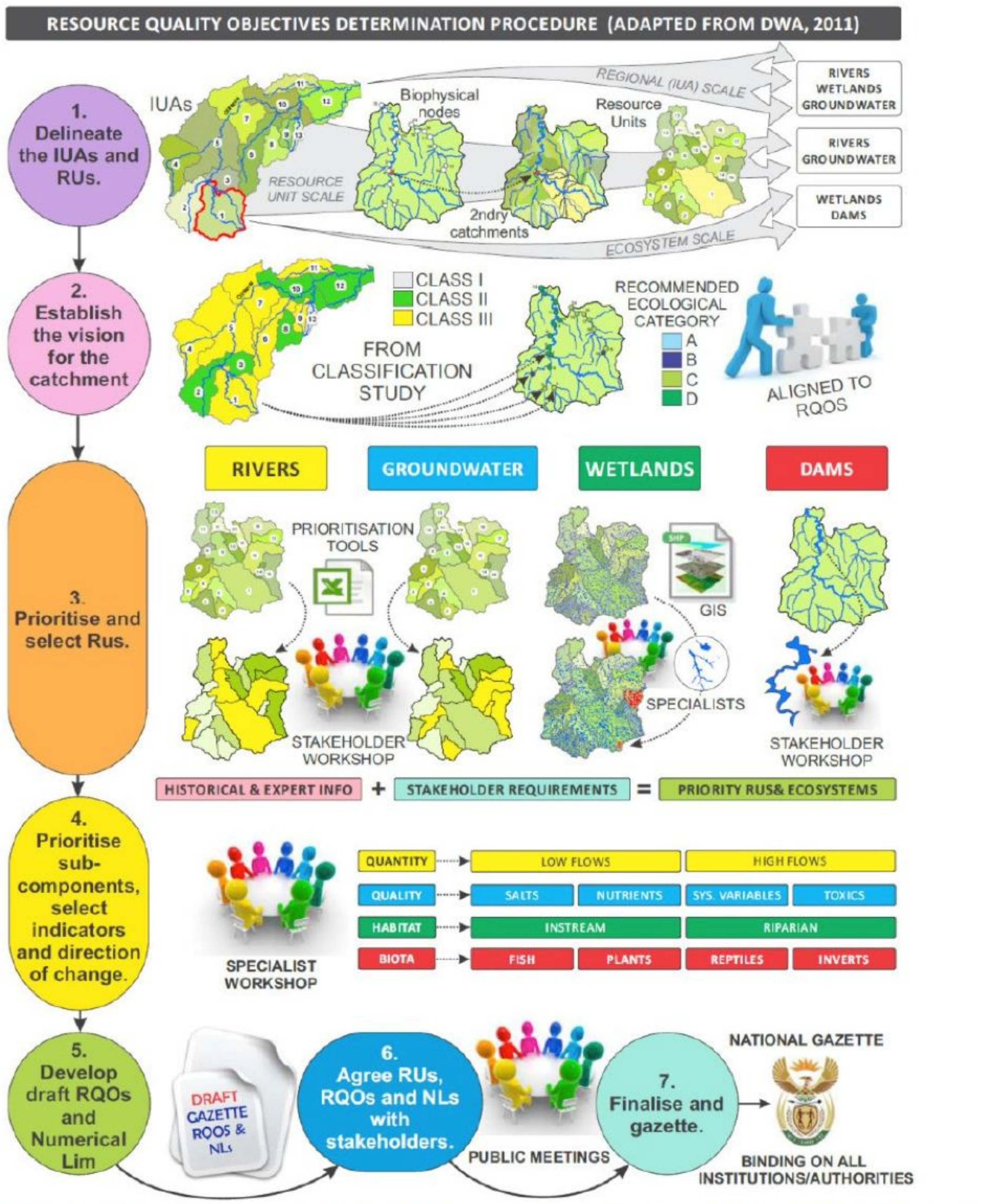


Figure 1: Schematic summary of the RQO determination procedure (adapted from DWA, 2011) which was implemented in this study.

National planning level application of the Olifants River results

The South African planning level review (DWA, 2011) contains the national generic Resource Water Quality Objectives for the country and lays out the adaptive management steps to improve water quality. At the national planning level, the water quality monitoring results for the Olifants WMA are presented in Table 5.10 (DWA, 2011) in terms of their degree of compliance with the Resource Quality Water Objectives (RWQOs) specified for it, alongside of those of the other WMAs assessed for planning purposes.

Each of the chemical variables monitored (viz. electrical conductivity (EC), sulphate, chloride, ortho-phosphate, ammonia, and pH) are assessed with regards their relative compliance with the RWQOs, in terms of the relative proportion of sites/water bodies that fall within the designated range limits, from ideal or acceptable (blue and green, respectively) to tolerable (yellow) or unacceptable (red). For instance, for electrical conductivity (EC), compliance was far weaker for the Olifants (43% of sites currently rated as unacceptable and only

14% as ideal) than for the Thukela WMA, where there were no sites at totally unacceptable EC levels, and the Mvoti-Mzimkulu, with 68% of EC values within the ideal range limits (see Table 5.10).

Table 5.10: Summary of water quality compliance to Resource Water Quality Objectives (RWQOs) for six parameters per Water Management Area (WMA). South Africa, for the monitoring sites assessed.
Source: DWA (2011).

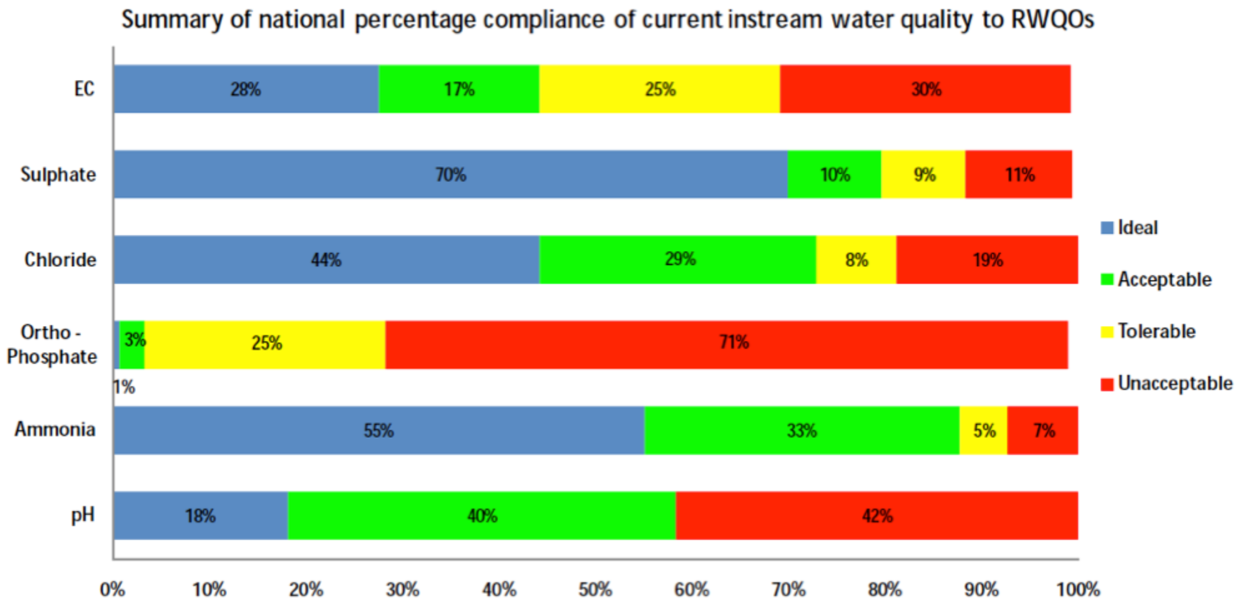
WMA	Electrical Conductivity (EC)	Sulphate (SO_4)	Chloride (Cl)	Ortho-phosphate (PO_4-P)	Ammonia (NH_3-N)	pH
1 - Limpopo	33% 17% 50%	17% 83%	50% 50%	17% 50% 33%	100%	17% 66% 17%
2 - Luvuvhu and Letaba	12% 44% 44%	100%	22% 33% 45%	44% 56%	11% 89%	11% 56% 33%
3 - Crocodile (West) and Marico	15% 62% 15% 8%	23% 77%	15% 46% 39%	69% 23% 8%	15% 8% 62% 15%	54% 46%
4 - Olifants	43% 36% 7% 14%	43% 7% 50%	14% 14% 21% 50%	36% 64%	64% 36%	57% 36% 7%
5 - Inkomati	7% 29% 14% 50%	7% 93%	29% 71%	43% 57%	14% 86%	29% 50% 21%
6 - Usustu to Mhlatuze	19% 25% 25% 31%	7% 7% 86%	19% 6% 19% 56%	50% 50%	6% 38% 56%	31% 31% 38%
7 - Thukela	10% 40% 50%	10% 90%	100%	80% 20%	10% 30% 60%	20% 60% 20%
8 - Upper Vaal	22% 34% 16% 28%	6% 22% 9% 63%	6% 34% 60%	91% 9%	15% 9% 38% 38%	53% 31% 16%
9 - Middle Vaal	50% 24% 13% 13%	13% 31% 6% 50%	19% 19% 38% 24%	100%	19% 12% 25% 44%	50% 44% 6%
10 - Lower Vaal	78% 22%	44% 44% 12%	11% 33% 56%	56% 44%	34% 44% 22%	78% 22%
11 - Mvoti to Mzimkulu	16% 16% 68%	100%	5% 26% 68%	32% 36% 21% 11%	5% 5% 32% 58%	16% 42% 42%
12 - Mzimvubu to Keiskamma	11% 20% 16% 53%	5% 95%	5% 11% 16% 68%	95% 5%	37% 63%	16% 79% 5%
13 - Upper Orange	16% 32% 32% 20%	5% 5% 90%	5% 32% 63%	84% 16%	16% 16% 68%	53% 47%
14 - Lower Orange	29% 29% 43%	100%	71% 29%	43% 57%	14% 14% 14% 57%	43% 57%
15 - Fish to Tsitsikamma	61% 18% 14% 7%	11% 18% 25% 46%	54% 7% 25% 14%	82% 18%	4% 7% 46% 43%	57% 29% 14%
16 - Gouritz	64% 18% 18%	35% 6% 18% 41%	64% 12% 12% 12%	94% 6%	6% 6% 24% 64%	47% 29% 24%
17 - Olifants Doorn	17% 17% 66%	17% 83%	17% 33% 50%	33% 67%	100%	50% 17% 33%
18 - Breede	72% 14% 14%	36% 21% 43%	72% 21% 7%	86% 14%	7% 7% 29% 57%	57% 14% 29%
19 - Berg	34% 22% 22% 22%	33% 67%	44% 44% 12%	100%	22% 78%	11% 22% 67%
Ideal range limit	30mS/m	80 mg/l	40 mg/l	0.005 mg/l	0.015 mg/l	≥6.5 - ≤8.0
Acceptable range limit	50 mS/m	165 mg/l	120 mg/l	0.015 mg/l	0.044 mg/l	>8.0 - ≤8.4
Tolerable range limit	85 mS/m	250 mg/l	175 mg/l	0.025 mg/l	0.073 mg/l	No range limit set
Unacceptable limit	> 85 mS/m	> 250 mg/l	> 175 mg/l	> 0.025 mg/l	> 0.073 mg/l	<6.5 and > 8.4

As illustrated in »Figure 5.27 below, when viewed in aggregate across WMAs, the monitoring results can be used to provide a snapshot at the national scale of the degree of compliance of present instream water quality to Resource Water Quality Objectives (RWQOs), for all of the key chemical constituents assessed (DWA, 2011).

The results of the water quality review highlighted the levels of nutrients in the country's water resources as the water quality problem of most concern, with fully 96% of sites assessed falling in the unacceptable to tolerable ranges for ortho-phosphate. Only 29% of the monitoring sites showed compliance to the prescribed RWQOs (≤ 0.025mg l-1) for phosphate; there is currently 71% of non-compliance at a national scale. This state was considered a threat to the aquatic ecosystem health of the country's water resources and to domestic water. Working back using the underlying data, it is possible to identify the WMA and specific water bodies and sites where different water quality issues are most problematic (as above). Similarly, salinisation was another major quality issue identified; 30% of the monitoring sites showed unacceptably high levels (> 85 mS m-1) of salts, and 25% fell within the tolerable range (50 mS m-1 to 85 mS m-1). Only 48 of the (17%) monitoring points assessed at a national scale met all the RWQOs for all water quality variables.

Based on the planning level review of water quality obtained in this way at a national scale and per WMA, a range of strategic water quality interventions were identified as DWA focus areas over the short to long term planning horizon, to achieve the objectives of resource directed water quality management (DWA, 2011). Significantly, the DWA (2011) review of the water quality information system for the country concluded that the DWAF (1996) WQGs (see »Chapter 3) are inadequate, for instance in terms of the limits in place for nutrients, toxics and salinity (see »Box 5.4); a set of new limits is tabulated in DWA (2011) for nitrogen, for example. The SA WQGs for aquatic ecosystems remain under refinement.

Figure 5.27: Degree of compliance of the present instream water quality to Resource Water Quality Objectives (RWQOs), by major chemical constituent, for South Africa. EC = Electrical Conductivity.
Source: DWA (2011).



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Example VII: The use of national data sets to assess present conditions of freshwater ecosystems in Mexico

Mexico provides an example of how readily available national data sets can be used to assess present condition, prioritise future management actions, and identify major data gaps at a large scale. Pressure on Mexico's water resources is intensifying, with population growth, increasing demand for water, food and energy, and the effects of climate change. The mega-biodiverse rivers of the country, which are vital for supporting socioeconomic growth, are deteriorating under this stress. Realizing this precipitated a national scale diagnosis of the current state of freshwater health in rivers and basins based on the methodology as detailed in Garrido Pérez et al. (2010a). A multi-criteria decision support model was developed, drawing on existing national scale data for 75 environmental indices used as surrogate measures of the extent of ecohydrological alteration in 393 river systems (»Figure 5.28). This assessment was conducted at three different scales: (1) river, (2) riparian zone and (3) entire basin. The indices were of varying degrees of national coverage, from widespread to patchy, and when aggregated into sub-model clusters addressed factors from water quality and urbanization, to groundwater exploitation.

The assessment showed that some 55% of river systems or parts thereof are already highly degraded. These impacted rivers presently support the wellbeing of almost 83 million people and cumulatively represent 313 000 river km and 49% of national territory. Of these, seven basins, which represent significant water resources for more than half the country's population, are already critically altered (»Figure 5.29). Encouraging, 224 primarily first and second order river basins, though representing only 14% of the country's area, remain relatively intact (e.g. small basins of Baja California Peninsula, endorheic basins in North-central México, and coastal rivers contributing flows to the Pacific Ocean and Gulf of México).

Figure 5.28: Conceptual model to evaluate the eco-hydrological alteration of rivers in Mexico.

Source: Adjusted from Garrido et al. (2010b).

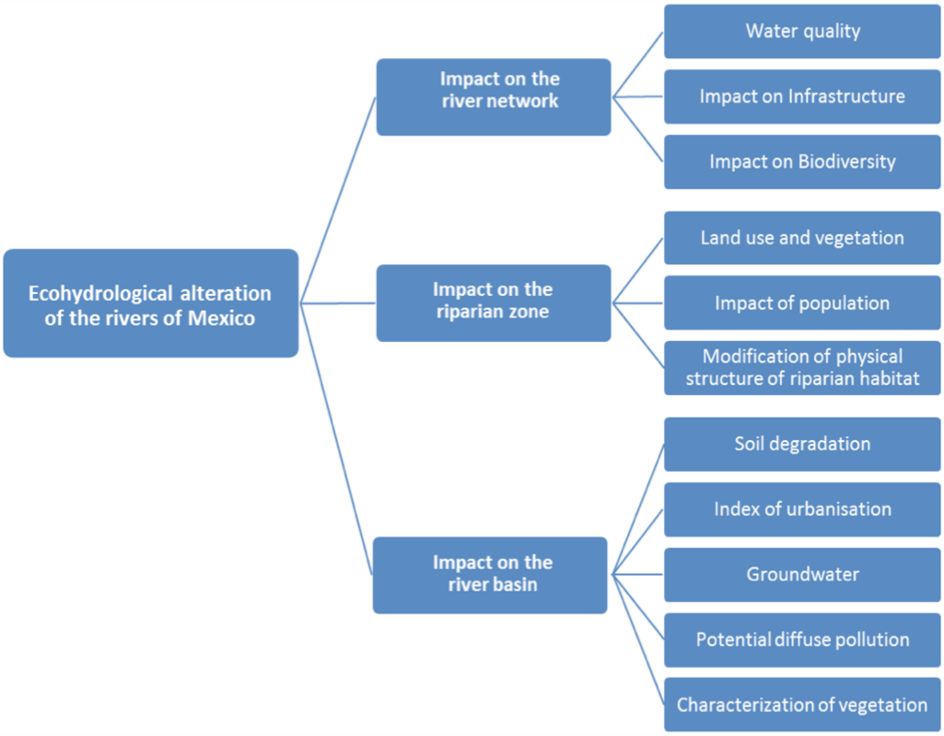

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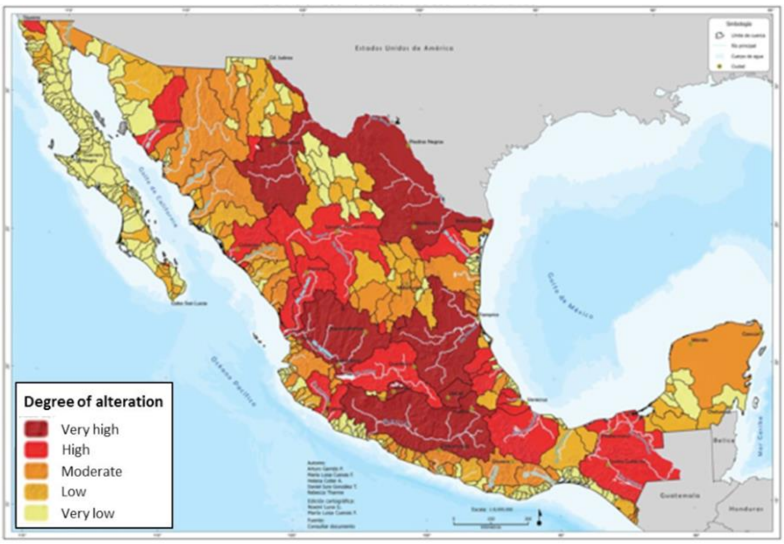
Figure 5.29: Alteration of the eco-hydrology of rivers in Mexico.

Source: Adapted from Garrido et al..(2010b).



Flow regime alteration and system fragmentation by water resources infrastructure, basin conversion of natural vegetation cover to other land uses, and the water quality effects of diffuse agricultural pollution, were found to be some of the influential drivers of river system decline. The results highlighted options for prioritizing national and basin strategies for river conservation, improved flow management, including through the provision of environmental flows, and for better land management practices (e.g. reduction of nutrient and sediment loads in agricultural field runoff.

6 Conclusions and recommendations for further development

Conclusions

The “4 phases – 9 steps” approach[65] as outlined and demonstrated in detail in Chapters 4 and 5 describes and illustrates a recurring cyclical process including the vision, formulation of guidelines and their implementation which includes monitoring, reporting and revisiting the objectives, assessing successes and weaknesses. It also includes the potentially necessary governance and institutional actions required to effectively support the process.

These cyclical, de facto perpetual activities reflect the never-ceasing concern about the state and potential impairment of freshwater bodies. The management and rehabilitation/safeguarding of freshwater ecosystems imply the necessity to revisit all 9 steps from time to time and to gradually improve the evidence base through increasingly tailor-made monitoring and evaluation. It is worth noting that one may start out with a vision and associated set of aspirational objectives and find that some of these are simply not achievable because of other constraints (e.g. economic development imperatives). In this case, it may be necessary to revisit the vision and objectives in a participatory process and see whether there is broad acceptance to modify these to meet more realistic expectations.

While it is difficult to estimate the duration of an “optimal cycle”, the necessary time span may be estimated to be between 5 to 10 years. Not all steps will have to be changed in every subsequent cycle. Water laws and other instruments of water governance, as shown in the practice of several countries in »Chapter 3, may have a lifetime of 20 to 30 years, thus about 2 or 3 cycles.

The framework for developing water quality guidelines for ecosystems outlines the different steps, from desktop studies to broad-scale monitoring to more local monitoring, and what can be expected as a result upon their completion. This stepwise approach, with a subsequently increasing degree of detail, facilitates the initiation of guideline and/or standard development and the rehabilitation efforts for freshwater ecosystems. In the context of the Sustainable Development Goals (SDGs), these alternative approaches advocated in this volume are in line with the spirit of the respective resolution of the United Nations General Assembly (UNGA) encapsulated in the statement “leaving no one behind”. The often cited “ladder approach” enabling developing countries to start with implementing and monitoring SDGs with a gradually improving set of methods and observations is also well reflected in the options provided in the versatile methodological recommendations of the “4 phases 9 steps” approach.

By proposing this staged approach, the Framework encourages further

[65] This 4 Phase, 9 Step Framework is the precursor of the 4 Phase, 12 Step Framework later adopted, as described in the preface section of this volume.

development once the initial assessment of the status of freshwater ecosystems is completed. It highlights the importance of these initial steps by showing how much has already been accomplished by following the ‘no further deterioration’ principle, the implementation of ‘no regret’ solutions, and through awareness raising.

Many restoration actions can be started on the basis of low cost assessments that build on publicly available and historical data. However, the use of Earth Observations and Big Data to also provide ever more and more detailed information on water quality, which can be updated, georeferenced and shared in real time, also provides exciting opportunities for future water quality monitoring, and these options should be further explored at a global scale.

Concerning the monitoring recommendations, the Framework refrains from maximalist requirements and rather argues for selecting fewer, yet relevant indicators, maintaining their monitoring sustainably. This advice implies of course that much care should be invested in the selection of these indicators (and even more in that of aggregated indices), since these indicators serve as the basis for important decisions towards improving the status of freshwater ecosystems. They will have to measure success and/or failure of restoration actions. Thus, their role and interpretation is not limited to a narrow scientific community of aquatic ecologists. Results of assessments should also be communicable to numerous stakeholders. Numerical values of indicators therefore have to be associated with targeted states of freshwater ecosystems and tangible, well explained benefits for people and nature alike.

Given the multitude and versatility of freshwater ecosystems, their functions and the threats they might be exposed to as well as the associated social and political objectives, it is extremely difficult to identify indicators and respective criteria or threshold values which would fit all cases globally. While indicators may be classified as physico-chemical, hydromorphological and biological, this volume makes explicitly clear that a comprehensive assessment of freshwater ecosystems has to rely substantially on biological monitoring and evaluation. Biological indicators are essential components of assessment, since they serve as “integrators” of multiple facets of the quality of life supporting attributes of freshwater ecosystems in space and time. However, it is neither easy nor cheap to monitor them. Therefore, improving capabilities and capacities for biological monitoring is likely to be the prime candidate for forming partnerships for joint monitoring, implementation and know-how transfer.

The substantial evidence of the United States Environmental Protection Authority (US EPA), the European Water Framework Directive (EU WFD) and other advanced guidelines with multiple attributes, multidimensional assessment and classification schemes show how much time, effort, costs, human and institutional capacities are needed to set up functioning, implementable and enforceable guidelines. Many countries in the world can and will not be able to afford to develop their own EU WFD-like or other advanced methodology unless they can secure external sources of funding and assistance. Simpler approaches do exist. However, the wisdom of Albert Einstein, who promoted that things should be as simple as possible but not simpler, is a powerful reminder that complex aquatic ecosystem functions and their assessment cannot be simplified beyond a certain limit. By presenting a state-of-the-art methodology, building on the latest scientific insights, the Framework subscribes to the principle that obligations towards ecosystem integrity, but also science and knowledge, are universal.

The SDGs clearly endorse the same principle by targeting the restoration and safeguarding of freshwater ecosystems worldwide. The global unity of the SDGs, obliging both developing and developed countries to achieve them, is also a powerful appeal to form sustainable partnerships, to engage in long-term assistance and international cooperation in achieving (among others) the water quality and freshwater ecosystem related targets.

The Framework is admittedly science based; however, it cannot be more advanced than the present level of universally available knowledge and scientific understanding. Consequently, the Framework reflects in its limitations the ongoing scientific debate related, for example, to the usefulness of universally binding water quality classifications or the nature, type and numerical values of (selected) indicators and indices to demarcate (to a certain extent arbitrary) quality categories or classes. In this context, the Framework attempts to follow a "middle path". »Annex 2 summarizes the most frequently used and recommended indicators and their numerical values, which might serve as benchmarks demarcating certain domains of the water quality/ecosystem health continuum.

The main emphasis of the Framework is to serve as an advisory manual on how to develop guidelines as described in Chapter 4. It cannot be overemphasized that this process should be seen as a policy and societal decision-making process as much as an objective scientific assessment.

The duality of addressing the human-centered utilitarian and well-being oriented, but also eco-centric aspects of the management and safeguarding of freshwater ecosystems is central to the paradigm of integrated water resources management. The emphasis on water as more than a simple resource, but also as an ecosystem and habitat is largely strengthened by the targets formulated for SDG Goal 6 by the Member States. This trend led to the modification of the internationally well accepted Drivers, Pressures, States, Impacts, Responses (DPSIR) model. The links between the anthropic and freshwater systems are captured by developing the concept of the pressures/stressors interface, hence the DPSSIR model as presented in »Section 2.42.4. In fact, human well-being and freshwater ecosystems are inseparable simply because the former relies on the services provided by the latter. However, the Framework is not only a guide to securing the provision of services of freshwater ecosystems; it also argues for considering the needs of freshwater ecosystems for their own sake. In this context, identifying high value refuge and conservation areas, saving freshwater biodiversity or identifying Key Biodiversity Areas (KBAs) goes well beyond the consideration of freshwater ecosystems as service providers.

Recommendations for further development

This volume has presented a Framework for developing Water Quality Guidelines, rather than a prescriptive Water Quality Guideline (WQG) (or standard) itself. In some parts and details, the Framework has the features of a manual, but its major role is to guide the development of context-specific, national WQGs. Follow-up activities after the present phase of the development of the Framework should assist their evolution in this direction.

This could include more emphasis on funding issues, providing details for capacity development curricula at several professional and technical levels, but also monitoring protocols, laboratory methods and equipment as well as advice on data archiving, reporting and public awareness raising. Further development of the Framework should also aim to assist countries to report on SDG indicators and achieve targets.

Thematically, this volume concentrates on inland surface waters with particular emphasis on freshwater ecosystems. However, groundwater aquifers are increasingly being recognized as ecosystems in their own right, though this ecosystem feature is yet to be explored in more scientific detail. Future development of the Framework could consider this potential extension.

Furthermore, major extensions can be envisaged for the coastal zone of intermittent brackish/freshwater ecosystems, interchange zones between salty aquifers and fresh surface waters, but also freshwater ecosystems of ephemeral nature (dry/wet in the hot zones, but also liquid/solid phases in cold regions). Likewise, the role and protection of wetlands is likely to be increasingly recognized. Specific guideline developments in these directions can be expected.

While the Framework address the question of environmental flows and their role in the sustenance of healthy freshwater ecosystems, this research field is currently developing rapidly. Consequently, it can be expected that, within a few years, substantial and new knowledge will be available to be considered in guidelines and standards.

There is no doubt that global environmental change, first and foremost climate change and increasing climate variability, but also human population dynamics with the welcome uplift of millions from poverty and the increase in the standard of living will require that the issues addressed by the Framework be revisited to take new challenges and stressors, but also new knowledge and science, into account.

Annex 1

References

Abdalla, C.W., Drohan, J.R., Becker, J.C. (2010). River Basin Approaches to Water .Management in the Mid-Atlantic States. The Pennsylvania State University, USA. http://pubs.cas.psu.edu/FreePubs/pdfs/ua466.pdf [Accessed 02 February 2016]

Abell, R., Thieme, M.L., Revenga, C., Bryer, M., Kottelat, M., Bogutskaya, N., Coad, B., Mandrak, N., Contreras Banderas, S., Bussing, W., Stiassny, M.L.J., Skelton, P., Allen, G.R., Unmack, P., Naseka, A., Ng, R., Sindorf, N., Robertson, J., Armijo, E., Higgins, J.V., Heibel, T.J., Wikramanayake, E., Olson, D., López, H.L., Reis, R.E., Lundberg, J.G., Sabaj Pérez, M.H., Petry, P. (2008). Freshwater ecoregions of the world: A new map of biogeographic units for freshwater biodiversity conservation. BioScience, 58, 403–414.

Abraham W., Macedo A.J., Gomes L.H., Tavares F.C.A. (2007). Occurrence and resistance of pathogenic bacteria along the Tietê River downstream of São Paulo in Brazil. Clean – Soil Air Water, 35, 339-347.

Acreman, M.C., Dunbar, M.J. (2004). Defining environmental flow requirements - a review. Hydrology and Earth System Sciences, 8, 861-876.

Acreman, M.C., Overton, I.C., King, J., Wood, P.J., Cowx, I.G., Dunbar, M.J., Kendy, E., Young, W.J. (2014). The changing role of ecohydrological science in guiding environmental flows. Hydrological Sciences Journal, 59(3-4), 433-450.

ACTFR (Australian Centre for Tropical Freshwater Research) (2004). A typological basis for the assessment and Management of wetland water quality in the dry and wet-dry regions of tropical Queensland. ACTFR Report No. 04/04.

Acuña, V., Datry, T., Marshall, J., Barceló, D., Dahm, C.N., Ginebreda, A., McGregor, G., Sabater, S., Tockner, K., Palmer, M.A. (2014). Why should we care about temporary waterways?. Science, 343, 1080-1081.

Adams, J.B. (2014). A review of methods and frameworks used to determine the environmental water requirements of estuaries. Hydrological Sciences Journal, 59(3-4), 451-465.

Adeoti, O. (2010). Development of River Organizations in Nigeria. Research Journal of Soil and Water Management, 1, 91-100.

Akbar, T.A., Hassan, Q.H., Achari, G. (2014). Development of Remote Sensing Based Models for Surface Water Quality. Clean – Soil, Air, Water, 42(8), 1044-1051.

Aldenberg T., Slob W. (1993). Confidence limits for hazardous concentrations based on logistically distributed NOEC toxicity data. Ecottoxicology and Environmental Safety, 25(1), 48-63.

Allan, J.D., Flecker, A.S. (1993). Biodiversity conservation in running waters. BioScience, 43 (1), 32-43.

Allan, T. (2003). IWRM/ WRAM: a new sanctioned discourse? School of Oriental and African Studies, Occasional, 1–27.

Allan, J.D. (2004). Landscapes and riverscapes: The influence of land use on stream ecosystems. Annual Review of Ecology Evolution and Systematics, 35, 257-284.

Allan, J.D., Abell, R., Hogan, Z., Revenga, C., Taylor, B.W., Welcomme, R.L., Winemiller, K. (2005). Overfishing of inland waters. BioScience, 55, 1041-1051.

Allen, A.P., Whittier, T.R., Larsen, D.P., Kaufmann, P.R., O'Connor, R.J., Hughes, R.M., Stemberger, R.S., Dixit, S.S., Brinkhurst, R.O., Herihy, A.T., Paulsen, S.G. (1999). Concordance of taxonomic composition patterns across multiple lake assemblages: effects of scale, body size and land use. Canadian Journal of Fisheries and Aquatic Sciences, 56, 2029-2040.

Allen, D.J., Molur, S., Daniel, B.A. (compilers) (2010). The status and distribution of freshwater biodiversity in the Eastern Himalaya. International Union for Conservation of Nature, Cambridge and Gland and Zoo Outreach Organisation, Coimbatore.

Allen, D.J., Smith, K.G., Darwall, W.R.T. (compilers) (2012). The status and distribution of freshwater biodiversity in Indo-Burma. International Union for Conservation of Nature, Cambridge and Gland.

Altenburger, R., Ait-Aissa, S., Antczak, P., Backhaus, T., Barceló, D., Seiler, T.B., Brion, F., Busch, W., Chipman, K., López de Alda, M., Umbuzeiro, G. A., Escher, B.I., Falciani, F., Faust, M., Focks, A., Hilscherova, K., Hollender, J., Hollert, H., Jäger, F., Jahnke, A., Kortenkamp, A., Krauss, M., Lemkine, G.F., Munthe, J., Neumann, S., Schymanski, E.L., Scrimshaw, M., Segner, H., Slobodnik, J., Smedes, F., Kughathas, S., Teodorovic, I., Tindall, A.J., Tollefsen, K.E., Walz, K.H., Williams, T.D., Van den Brink, P.J., van Gils, J., Vrana, B., Zhang, X., Brack, W. (2015). Future water quality monitoring — Adapting tools to deal with mixtures of pollutants in water resource management. Science of the Total Environment, Volumes 512–513, 540–551.

ANA (National Water Agency of Brazil) (2012). Surface Freshwater Quality in Brazil – Outlook (2012). Executive Summary. AgênciaNacional de Águas, Brasília.

Anderson, K., Gaston, K. J. (2013). Lightweight unmanned aerial vehicles will revolutionize spatial ecology. Frontiers in Ecology and the Environment, 11(3), 138-146.

Anderson, D. M., Cembella, A. D., Hallegraeff, G. M. (2012). Progress in understanding harmful algal blooms: paradigm shifts and new technologies for research, monitoring, and management. Annual review of marine science, 4, 143-176.

Annear T., Chisholm I., Beecher H., Locke A., Aarrestad P., Burkhard N., Coomer C., Estes C., Hunt J., Jacobson R., Jobsis G., Kauffman J., Marshall J., Mayes K., Wentworth R., Stalnaker C. (2004). Instream Flows for Riverine Resource Stewardship, revised edition. Instream Flow Council, Cheyenne, WY.

Anonymous. (1995). Sustaining the World's Forests: The Santiago Agreement. Journal of Forestry, 93, 18-21.

ANZECC/ARMCANZ (Australian and New Zealand Environment and Conservation Council/ Agriculture and Resource Management Council of Australia and New Zealand) (2000a). Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Volume 1, The Guidelines (chapters 1-7), Australian and New Zealand Environment and Conservation Council. http://pubs.cas.psu.edu/FreePubs/pdfs/ua466.pdf [Accessed 02 February 2016] http://www.environment.gov.au/resource/australian-and-new-zealand- guidelines-fresh-and-marine-water-quality-volume-1-guidelines

ANZECC/ARMCANZ (Australian and New Zealand Environment and Conservation Council/ Agriculture and Resource Management Council of Australia and New Zealand) (2000b). Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Volume 2, Aquatic Ecosystems – Rationale and Background Information (chapter 8), Australian and New Zealand Environment and Conservation Council. http://www.environment.gov.au/topics/water/publications-and-resources/water-quality [Accessed 02 February 2016]

ANZECC/ARMCANZ (Australian and New Zealand Environment and Conservation Council/ Agriculture and Resource Management Council of Australia and New Zealand) (2000c). Australian and New Zealand Guidelines for Fresh and Marine Water Quality, Volume 2, Primary Industries – Rationale and Background Information. (Chapter 9), Australian and New Zealand Environment and Conservation Council. http://www.environment.gov.au/topics/water/publications-and-resources/water-quality [Accessed 02 February 2016]

Apip, M., Sulastri, F.,Subehi, L., Ridwansyah,I. (2003). Assessing Climate Elements in physical process of Lake Maninjau. LIMNOTEK, 10(1), 1-10.

Aquatic Ecosystems Task Group (2012). Aquatic Ecosystems Toolkit. Module 1: Aquatic Ecosystems Toolkit Guidance Paper. Australian Government Department of Sustainability, Environment, Water, Population and Communities, Canberra. http://www.environment.gov.au/water [Accessed 02 February 2016]

Aragón-Noriega, L. E. & Calderon-Aguilera, E. A. (2000). Does damming of the Colorado River affect the nursery area of blue shrimp Litopenaeus stylirostris (Decapoda: Penaeidae) in the Upper Gulf of California?. Revista de Biologia Tropical, 48(4), 867-871;

Arismendi, I., Johnson, S. L., Dunham, J. B., Haggerty, R. (2013). Descriptors of natural thermal regimes in streams and their responsiveness to change in the Pacific Northwest of North America. Freshwater Biology, 58(5), 880-894.

Aroviita, J., Mykra, H., Muotka, T., Hämäläinen, H. (2009). Influence of geographical extent on typology- and model-based assessments of taxonomic completeness of river macroinvertebrates. Freshwater Biology, 54, 1774–87.

Arthington, A.H., Bunn, S.E., Poff, N.L., Naiman, R.J. (2006). The challenge of providing environmental flow rules to sustain river ecosytems. Ecological Applications, 16(4), 1311–1318.

Arthington, A.H. (2012). Environmental Flows: Saving Rivers in the Third Millennium. Freshwater Ecology Series. University of California Press, Berkeley and Los Angeles, California, and London, England.

Ashander, J., Krkosek, M., Lewis M. (2012) Aquaculture-induced changes to dynamics of a migratory host and specialist parasite: a case study of pink salmon and sea lice. Theoretical Ecology, 5(2), 231-252.

Ash, N., Blanco, H., Brown, C., Garcia, K., Henrichs, T., Lucas, N., Raudsepp-Hearne, C., Simpson, R.D., Scholes, R., Tomich, T.P., Vira, B., Zurek, M. (2010). Ecosystems and Human Well-Being. A Manual for Assessment Pratitioners. Island Press, Washington DC.

Aubin, D., Varone, F. (2004). The Evolution of European Water Policy. In: Kissling-Näf, I., Kuks, S. (eds.). The Evoluation of National Water Regimes in Europe. Transitions in Water Rights and Water Policies. Kluwer Academic Publishers, 49-86.

Austin M.P., Smith T.M. (1989). A new model for the continuum concept. Vegetatio, 83, 35-47.

Ayres, I., Braithwaite, J. (1992). "Responsive Regulation: Transcending the deregulation debate". Oxford University Press, New York.

Ayers, R.S., Westcot, D.W. (1985). Water quality for agriculture (Vol. 29). Food and Agriculture Organization of the United Nations, Rome.

Bailey, R.C., Norris, R.H., Reynoldson, T.B. (2004). Bioassessment of Freshwater Ecosystems: Using the Reference Condition Approach. Kluwer Academic Publishers, New York.

Bailey, R.C., Norris, R.H., Reynoldson, T.B. (2001). Taxonomic resolution of benthic macroinvertebrate communities in bioassessments. Journal of the North American Benthological Society, 20, 280–286.

Balian, E.V., Lévêque, C., Segers, H., Martens, K. (eds.) (2008). Freshwater Animal Diversity Assessment. Springer Verlag, Heidelberg.

Baskett, M.L., Burgess, S.C., Waples, R.S. (2013). Assessing strategies to minimize unintended fitness consequences of aquaculture on wild population. Evolutionary Applications, 6(7), 1090-1108.

Barber, M.C. (1994). Environmental Monitoring and Assessment Program: Indicator Development Strategy. EPA/620/R-94/022. US Environmental Protection Agency, Office of Research and Development, Environmental Research Lab, Athens, GA.

Barbour, M.T., Gerritsen, J., Snyder, B.D., Stribling, J.B. (1999). Rapid bioassessment protocols for use in streams and wadeable rivers. US Environmental Protection Agency, Washington, D.C.

Barbour, M. T., Yoder, C.O. (2000). The multimetric approach to bioassessment in the United States of America. In: Wright, J.F., Sutcliffe, D.W. Ambleside, F.M.T. (eds.). Assessing the biological quality of freshwaters. RIVPACS and other techniques. Freshwater Biological Association, 281-292.

Barbour, M.T., Gerritsen, J. (2006). Key features of bio assessment development in the United States of America. In: Ziglio, G., Siligardi, M., Flaim, G.(eds.). Biological Monitoring of Rivers: Applications and Perspectives. John Wiley & Sons Ltd., 351–366.

Barbour, M.T., Stribling, J.B., Verdonschot, P.F.M. (2006). The multihabitat approach of USEPA's rapid bioassessment protocols: benthic macroinvertebrates, Limnetica, 839-850.

Barella W., Petrere M. Jr. (2003). Fish community alterations due to pollution and damming in Tietê and Paranapanema rivers (Brazil). River Research and Applications 19: 59–76.

Barrick Porgera (2012). Environmental Monitoring Annual Report 2012. http://www.barrick.com/files/porgera/2012-Porgera-Annual-Environmental-Report.pdf [Accessed 18 February 2016]

Barrios, V., Carrizosa, S., Darwall, W. R. T., Freyhof, J., Numa, C., & Smith, K. (2014). Freshwater Key Biodiversity Areas in the Mediterranean Basin Hotspot: Informing species conservation and development planning in freshwater ecosystems (Vol. 52). IUCN, Gland. Cambridge and Malaga.

Bartram, J., Balance, R. (1996). Water Quality Monitoring. A Practical Guide to the Design and Implementation of Freshwater Quality Studies and Monitoring Programmes. Published on behalf of UNEP and WHO. E & FN Spon, London, pp. 383.

Bear, E.A., McMahon, T.E., Zale, A.V. (2007). Comparative thermal requirements of westslope cutthroat trout and rainbow trout: Implications for species interactions and development of thermal protection standards. Transactions of the American Fisheries Society, 136, 1113–21.

Belletti, B., Rinaldi, M., Buijse, A. D., Gurnell, A. M., Mosselman, E. (2014). A review of assessment methods for river hydromorphology. Environmental Earth Sciences, 73(5), 2079-2100.

Bellucci, C., Beauchene M., Becker M. (2008). Physical, Chemical, and biological Attributes of Moderately Developed Watersheds within Connecticut. http://www.ct.gov/deep/lib/deep/water/water_quality_management/ic_studies/mod_ic_rpt.pdf [Accessed 02 February 2016]

Bennion, H., Fluin, J., Simson, G.L., (2004). Assessing eutrophication and reference conditions for Scottish freshwater lochs using subfossil diatoms. Journal of Applied Ecology, 41, 124-138.

Benoy, G.A., Sutherland, A.B., Culp, J.M., Brua, R.B. (2010). Physical and ecological thresholds for deposited sediemtns in streams in agricultural landscapes, Special Section. Environmental Standards for Agricultural Watersheds, Journal of Environmental Quality. 41, 31-40.

Bernhardt, E., Bunn, S.E., Hart, D.D., Malmqvist, B., Muotka, T., Naiman, R.J., Pringle, C., Reuss M., van Wilgen, B. (2006). Perspective: The challenge of ecologically sustainable water management. Water Policy, 8, 475–479.

Biggs, J., Williams, P., Whitfield, M., Nicolet, P., Weatherby A. (2005). 15 years of pond assessment in Britain: Results and lessons learned from the work of Pond Conservation. Aquatic Conservation: Marine and Freshwater Ecosystems, 15, 693-714.

BioFresh. (2016). BioFresh Project. http://www.freshwaterbiodiversity.eu/. [Accessed 18 February 2016]

Birk S., Bonne, W., Borja, A., Brucet, S., Courrat, A., Poikane, S., Solimini, A., van de Bund, W., Zampoukas, N., Hering, D.I. (2012). Three hundred ways to assess Europe's surface waters: An almost complete overview of biological methods to implement the Water Framework Directive. Ecological Indicators, 18, 31–41.

Bizzi, S., Demarchi, L., Grabowski, R. C., Weissteiner, C. J., van de Bund, W. (2015). The use of remote sensing to characterise hydromorphological properties of European rivers. Aquatic Sciences, 1-14.

Blocksom, K.A., Kurtenbach, J.P., Klemm, D.J., Fulk, F.A., Cormier, S.M. (2002). Development and evaluation of the Lake Macroinvertebrate Inegrity Index (LMII) for New Jersey Lakes and Reservoirs. Environmental Monitoring and Assessment, 77, 311-333.

Bogardi, J.J., Pahl-Wostl, C., Vörösmarty, C. (2011). Water Security: Challenges for Science and Policy – Interconnected problems of a changing world call for sustainable solutions. (Blue Note). Global Water System Project, Bonn.

Bogardi, J.J., Dudgeon, D., Lawford, R., Flinkerbusch, E., Meyn, A., Pahl-Wostl, C., Vielhauer, K., Vörösmarty, C. (2012). Water security for a planet under pressure: interconnected challenges of a changing world call for sustainable solutions Elsevier Current opinion in Environmental Sustainability, 4(1), 35-43.

Bonada, N., Prat, N., Resh, V.H., Statzner, B. (2006). Developments in aquatic insect biomonitoring: A comparative analysis of recent approaches. Annual Review of Entomology, 51, 495-523.

Boon, P.J., Holmes, N.T.H., Raven, P.J. (2010). Developing Standard Approaches for Recording and Assessing River Hydromorphology: The Role of the European Committee for Standardization (CEN). Aquatic Conservation: Marine and Freshwater Ecosystems, 20, 55-61.

Booth, C.A., Lamble, P. (2012). Improving Water Quality- Stories of progress and success from across Australia. https://www.environment.gov.au/system/files/resources/4b9b3589-c2eb-4b51-aa7a-e9ab554d76af/files/improving-water-quality-section-1.pdf [Accessed 02 February 2016]

Borja, A., Josefson, A.B., Miles, A., Muxika, I., Olsgard, F., Phillips, G., Rodríguez, G., Rygg, B. (2007). An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive. Marine Pollution Bulletin, 55, 42–52.

Borja, A., Dauer, D.M. (2008). Assessing the environmental quality status in estuarine and coastal systems: Comparing methodologies and indices. Ecological Indicators, 8, 331-337.

Borja, A., Miles, A., Occhipinti-Ambrogi, A., Berg, T. (2009). Current status of macroinvertebrate methods used for assessing the quality of European marine waters: implementing the Water Framework Directive. Hydrobiologia, 633(1),181–96.

Bozkir, E.D., Leentvaar, J., Frijters, I.D., Hofstra, M.A. (2010). Impacts of National and International Actors on River Basin Management – Case of River Rhine. In: The Global Dimensions of Change in River Basins. Global Water System Project, Bonn, UNESCO-IHE, Delft, 125 - 139.

Brazil (1997). Law no 9433. The National Water Resource Policy.

Breine, J.J., Maes, J., Quataert, P., Van den Bergh, E., Simoens, I., van Thuyne, G., Belpaire, C. (2007). A fish-based assessment tool for the ecological quality of the brackish Schelde estuary in Flanders (Belgium). Hydrobiologia, 575, 141- 159.

Brierley, G., Hillman, M., Fryirs, K. (2006). Knowing your place: An Australasian perspective on catchment-framed approaches to river repair. Australian Geographer, 37(2), 131-145.

Brinson, M.M. (1993). A Hydrogeomorphic classification for wetlands. Wetlands Research Program Technical Report WRP-DE-4. US Army Corps of Engineers, Washington, DC.

Brisbane Declaration (2007). The Brisbane Declaration of the International Riversymposium and International Environmental Flows conference. Brisbane, Australia. Available at http://www.eflownet.org/download_documents/brisbane-declaration-english.pdf [Accessed 02 February 2016]

Brown, C.A., Joubert, A.R., Beuster, J., Greyling A., King, J.M. (2013). DRIFT: DSS software development for Integrated Flow Assessments. Final report. February 2013. Report to the Water Research Commission. WRC project No.: K5/1873.

Buck, O., Dev, K., Niyogi, D.K., Townsend C.R. (2004). Scale-dependence of land use effects on water quality of streams in agricultural catchments. Environmental Pollution, 130, 287–299.

Bunn, S.E., Davies, P.M., Mosisch, T.D. (1999). Ecosystem measures of river health and their response to riparian and catchment degradation. Freshwater Biology 41, 333-345.

Bunn, S.E., Davies, P.M. (2000). Biological processes in running waters and their implications for the assessment of ecological integrity. Hydrobiologia, 422/423, 61–70.

Bunn, S.E., Arthington, A.H. (2002). Basic principles and consequences of altered hydrological regimes for aquatic biodiversity. Environmental Management 30, 492–507.

Bunn S.E. (2003) Healthy River Ecosystems: vision or reality? Water, 30, 7–11.

Bunn, S.E., Apte, S.C., Bentley, K., Hart, B.T., Smith, R.E.W. (2006) .Porgera Riverine Monitoring Program Review and Optimisation. Porgera Joint Venture, PNG.

Bunn, S.E., Abal, E.G., Smith, M.J., Choy, S.C., Fellows, C.S., Harch, B.D., Kennard, M.J., Sheldon, F. (2010). Integration of science and monitoring of river ecosystem health to guide investments in catchment protection and rehabilitation. Freshwater Biology, 55(s1), 223-240.

Busch, M., La Notte, A., Laporte, V., Erhard, M. (2012). Potentials of quantitative and qualitative approaches to assessing ecosystem services. Ecological Indicators, 21, 89–103.

Butchart, S.H.M., Walpole, M., Collen, B., von Strien, A., Scharlemann, J.P.W., Almond, R.E.A., Baillie, J.E.M., Bomhard, B., Brown, C., Bruno, J., Carpenter, K.E., Carr, G.M., Chanson, J., Chenery, A.M., Csirke, J., Davidson, N.C., Dentener, F., Foster, M., Galli, A., Galloway, J.N., Genovesi, P., Gregory, R.D., Hockings, M., Kapos, V., Lamarque, J.-F., Leverington, F., Loh, J., McGeoch, M.A., McRae, L., Minasyan, A., Hernández Morcillo, M., Oldfield, T.E.E., Pauly, D., Quader, S., Revenga, C., Sauer, J.R., Skolnik, B., Spear, D., Stanwell – Smith, D., Stuart, S.N., Symes, A., Tierney, M., Tyrrell, T.D., Vié, J.-C., Watson, R., (2010). Global Biodiversity: Indicators of Recent Declines. Science, 328, 1164-1168.

Byström, P., Persson, L., Wahlstrom, E. (1998). Competing predators and prey: juvenile bottlenecks in whole-lake experiments. Ecology, 79, 2153–2167.

CABRI-Volga (2006) Environmental Risk Management in the Volga Basin and EU River Basins. State-of-the-Art Report – Executive Summary. CABRI-Volga project, www.cabri-volga.org. [Accessed 02 February 2016]

Cairns, J. Jr., McCormick, P.V., Niederlehner, B.R. (1993). A proposed framework for developing indicators of ecosystem health. Hydrobiologia, 263, 1-44.

Cairns, J., Pratt, J.R. (1993). A history of biological monitoring using benthic macroinvertebrates. In: Rosenberg, D.M., Resh, V.H. (eds.). Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, New York, 10-27.

Camargo, J.A., Alonso, A. (2006). Ecological and toxicological effects of inorganic nitrogen pollution in aquatic ecosystems: A global assessment. Review article in: Environment International, 32,831–849. http://www.uah.es/universidad/ecocampus/docs/6.pdf [Accessed 18 February 2016]

Canada Water Act (1985). Canada water Act, R.S.C, 1985, c. C11. Ministry of Justice, Canada. http://laws-lois.justice.gc.ca [Accessed 02 February 2016]

Cao, Y., Williams, D.D., Williams, N.E. (1998). How important are rare species in aquatic community ecology and bioassessment?. Limnology and Oceanography, 43, 1401-1409.

Cardoso, A., Free, G. Nõges, P.. Kaste, Ø., Poikane, S., Lyche Solheim, A. (2009). Lake Management, Criteria. In: Lake Ecosystem Ecology, G.E. Likens, ed.. Academic Press, Elsevier Inc. Amsterdam.

Cardoso, P., Erwin, T.L., Borges, P.A.V., New, T.R. (2011). The seven impediments in invertebrate conservation and how to overcome them. Biological Conservation, 144, 2647-2655.

Carletti, A.,Heiskanen, A.S. (2009). Water Framework Directive intercalibration technical report. Part 3: Coastal and Transitional waters. Joint Research Centre Institute for Environment and Sustainability, European Commission. http://publications.jrc.ec.europa.eu/repository/bitstream/111111111/10473/1/3010_08-volumecoast.pdf [Accessed 18 February 2016]

Carstensen, J. (2007). Statistical principles for ecological status classification of Water Framework Directive monitoring data. Marine Pollution Bulletin, 55, 3-15.

Carstensen, L., Henriksen, P. (2009). Phytoplankton biomass to nitrogen inputs: a method for WFD boundary setting applied to Danish coastal waters. Hydrobiologia, 633, 137-149.

Carvalho, L., Solimini, A., Phillips G., Pietiläinen, O-P., Mor, J., Cardosa, A.C., Lyche-Solheim, A., Ott, I., Søndergaard, M., Tartari, G., Rekolainen, S. (2009). Site-specific chlorophyll reference conditions for lakes in Northern and Western Europe. Hydrobiologia, 633, 59–66.

Cassidy, R., Jordan, P. (2011). Limitations of instantaneous water quality sampling in surface-water catchments: Comparison with near-continuous phosphorus time-series data. Journal of Hydrology, 405, 182-193.

Castello, L., Viana, J.P., Watkins, G., Pinedo-Vasquez, M., Luzadis, V. A. (2009). Lessons from Integrating Fishers of Arapaima in Small-Scale Fisheries Management at the Mamiraua Reserve, Amazon. Environmental Management, 43, 197-209.

Castells, M. (2008). The New Public Sphere: Global Civil Society, Communication Networks, and Global Governance. The ANNALS of the American Academy of Political and Social Science, 616(1),78–93.

Caux, P.-Y., Moore, D.R.J., MacDonald, D. (1997). Ambient Water Quality Guidelines (Criteria) for turbidity, Suspended and Benthic Sedimments, Technical appendix, Prepared for the BC Ministry of Environment, Lands, and Parks. http://www.env.gov.bc.ca/wat/wq/BCguidelines/turbidity/turbiditytech.pdf [Accessed 02 February 2016]

CCME (Canadian Council of Ministers of the Environment) (1995). Protocol for the derivation of Canadian sediment quality guidelines for the protection of aquatic life. CCME EPC-98E. Prepared by Environment Canada, Guidelines Division, Technical Secretariat of the CCME Task Group on Water Quality Guidelines, Ottawa. [Reprinted in Canadian environmental quality guidelines, Chapter 6, Canadian Council of Ministers of the Environment, 1999, Winnipeg.]

CCME (Canadian Council of Ministers of the Environment) (1999). Canadian Water Quality Guidelines for the Protection of Aquatic Life, Dissolved Oxygen. http://ceqg-rcqe.ccme.ca/download/en/177 [Accessed 02 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2004). Canadian Water Quality Guidelines for the protection of Aquatic Life. Phosphorus: Canadian framework for the management of freshwater systems. http://ceqg-rcqe.ccme.ca/download/en/205 [Accessed 02 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2007). A Protocol for the Derivation of Water Quality Guidelines for the Protection of Aquatic Life. In: Canadian Environmental Quality Guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg.

CCME (Canadian Council of Ministers of the Environment) (2010). Canadian Water Quality Guidelines for the Protection of Aquatic Life, Ammonium. http://ceqg-rcqe.ccme.ca/download/en/141 [Accessed 02 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2012). Canadian Water Quality Guidelines for the Protection of Aquatic Life. Nitrate Ion. http://ceqg-rcqe.ccme.ca/download/en/197 [Accessed 02 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2014a). Canadian Environmental Quality Guidelines, Summary table. [Accessed 18 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2014b). Canadian Water Quality Guidelines for the Protection of Aquatic Life, Fact Sheets. Canadian Council of Ministers of the Environment. http://ceqg-rcqe.ccme.ca/en/index.html#void [Accessed 18 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2014c). Canadian Water Quality Guidelines for the Protection of Aquatic Life, Water Quality Index. CCME WATER QUALITY INDEX 1.0 User's Manual Canadian Council of Ministers of the Environment. http://www.ccme.ca/files/Resources/calculators/WQI%20User's%20Manual%20(en).pdf [Accessed 18 February 2016]

CCME (Canadian Council of Ministers of the Environment) (2016).Water Quality Guidelines for the Protection of Aquatic Life. http://st-ts.ccme.ca/en/index.html [Accessed 09 March 2016]

CETESB (Sao Paulo Environmental Agency) (2008). Acompanhamento das indústrias da fase I do Projeto do Tietê. Companhia Ambiental do Estado de São Paulo. São Paulo, 2008.

CETESB (Sao Paulo Environmental Agency) (2013). Apendice C, Indices da Qualidade das Águas. http://aguasinteriores.cetesb.sp.gov.br/wp-content/uploads/sites/32/2013/11/Ap%C3%AAndice-C-%C3%8Dndices-de-Qualidade-das-%C3%81guas.pdf [Accessed 18 February 2016]

CETESB (Sao Paulo Environmental Agency) (2014). Qualidade das águas superficiais no Estado de São Paulo 2013. Companhia Ambiental do Estado de São Paulo, São Paulo.

Chan, K.M.A., Satterfield, T., Goldstein, J. (2012). Rethinking ecosystem services to better address and navigate cultural values. Ecological Economics, 74, 8-18.

Chapman. P. M. (1990). The sediment quality triad approach to determining pollution-induced degradation. Science of the Total Environment. 97-98, 815-825.

Chapman, P.M. (1995). Sediment quality assessment: Status and outlook. Journal of Aquatic Ecosystem Health, 4, 183–194.

Chapman, L.J., Chapman, C.A., Kaufman, L., Witte, F., Balirwa, J. (2008). Biodiversity conservation in African inland waters: Lessons of the Lake Victoria region. Verhandlungen des Internationalen Verein Limnologie, 30, 16-34.

Chase, M.C., Leibold, M.A. (2002). Spatial scale dictates the productivity-biodiversity relationship. Nature, 416, 427-430.

Cheethan, G., Chivers, G. (1996). Towards a holistic mdel of professional competence. Journal of European Industrial Training, 20, 20-30.

Cheethan, G., Chivers, G. (2005). Professions, competence and informal learning. Edward Elgar Publishing, Cheltenham.

China (Ministry of Environmental Protection of China) (2002). Environmental quality standards for surface waters. Document GB 3838-2002. 2002-06-01. http://english.mep.gov.cn/standards_reports/standards/water_environment/quality_standard/200710/t20071024_111792.htm [Accessed 18 February 2016]

Clark, M.J. (2002). Dealing with uncertainty: adaptive approaches to sustainable river management. Aquatic Conservation: Marine and Freshwater Ecosystems, 12, 347-363.

Clark, J. S. (2003). Uncertainty in Ecological Inference and Forecasting. Ecology, 84, 1349–1350.

Clarke, R. (2000). Uncertainty in estimates of biological quality based on RIVPACS. In Wright, J.F., Sutcliffe, D.W., Furse M.T. (eds). Assessing the biological quality of fresh waters: RIVPACS and other techniques. Freshwater Biological Association, Ambleside, Cumbria.

Clarke, R.T., Furse, M.T., Gunn, R.J.M., Winder, J.M., Wright, J.F. (2002). Sampling variation I in macroinvertebrate data and implications for river quality indices. Freshwater Biology, 47, 1735-1751.

Clarke, R.T., Wright, J.F., Furse, M.T. (2003). RIVPACS models for predicting the expected macroinvertebrate fauna and assessing the ecological quality of rivers. Ecological Modelling, 160, 219–33.

Clarke R.T., Hering D. (2006). Errors and uncertainty in bioassessment methods - major results and conclusions from the STAR project and their application using STARBUGS. Hydrobiologia, 566, 433-439.

Clarke R.T. (2010). WISER Deliverable D6.1-3: WISERBUGS (WISER Bioassessment Uncertainty Guidance Software) tool for assessing confidence of WFD ecological status class User Manual and software: Release 1.1 (Sept 2010). Technical Report. European Union FP7 Project WISER.

Clements, W. H., Kotalik, C. (2014). Effects of major ions on benthic communities: an experimental test of the proposed US EPA aquatic life benchmark for conductivity. Society of Environmental Toxicology and Chemistry. Vancouver, BC, Canada.

COAG (Council of Australian Governments) (2009). National Framework for Compliance and Enforcement Systems for Water Resource Management. http://www.environment.gov.au/resource/national-framework-compliance-and-enforcement-systems-water-resource-management [Accessed 02 February 2016]

COAG (Council of Australian Governments) (2012). National Framework for Compliance and Enforcement Systems for Water Resource Management. http://www.environment.gov.au/resource/national-framework-compliance-and-enforcement-systems-water-resource-management http://www.environment.gov.au/resource/national-framework-compliance-and-enforcement-systems-water-resource-management [Accessed 02 February 2016]

Coe, C.K. (2003). A report card on report cards. Public Performance and Management Review, 27, 53–76.

Cole, J.J., Prairie, Y.T., Caraco, N.F., McDowell, W.H., Tranvik, L.J., Striegl, R.G., Duarte, C.M., Kortelainen, P., Downing, J.A., Middelburg, J.J., Melack, J., (2007). Plumbing the global carbon cycle: Integrating inland waters into the terrestrial carbon budget. Ecosystems, 10(1), 171-184.

Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E., Cumberlidge, N., Darwall, W. R., Pollock, C., Richman, N.I., Soulsby, A-M., Böhm, M. (2014). Global patterns of freshwater species diversity, threat and endemism. Global Ecology and Biogeography, 23(1), 40-51.

Colombia (1984). Decreto 1594 de 1984. Usos del agua y residuos liquidos. (Spanish). http://www.alcaldiabogota.gov.co/sisjur/normas/Norma1.jsp?i=18617 [Accessed 02 February 2016]

Colombia (2010). Decreto numero 3930 de 25 October 2010, Ministerio de Ambiente, vivienda y desarrollo territorial. http://www.alcaldiabogota.gov.co/sisjur/normas/Norma1.jsp?i=40620 [Accessed 02 February 2016]

Conn, D.B. (2014) Aquatic invasive species and emerging infectious disease threats: A One Health perspective. Aquatic Invasions, 9,(3), 383-390.

Cook, C., Bakker, K. (2012). Water Security: Debating an emerging paradigm. Global Environmental Change, 22, 94-102.

Copeland, C. (2010). Clean Water Act. A Summary of the Law. Report No. 30030, Congressional Service, Washington, DC. http://www.in.gov/idem/files/rules_erb_20130213_cwa_summary.pdf [Accessed 02 February 2016]

Corcoran, E., Nellemann, C., Baker, E., Bos, R., Osborn, D., Savelli H. (eds.) (2010). Sick Water? The central role of wastewater management in sustainable development. A Rapid Response Assessment. United Nations Environment Programme, UN-HABITAT, GRID-Arendal. http://www.grida.no/publications/rr/sickwater/ [Accessed 02 February 2016]

Coria, J., Sterner, T. (2011). Natural Resource Management: Challenges and Policy Options (October 2011). Annual Review of Resource Economics, 3(1), 203-230.

Cormier, S.M., Norton, S.B., Suter, G.W. (2003). The US Environmental Protection Agency`s Stressor Identification Guidance : A process for determining the probable causes of biological impairments. Human and ecologicail Risk Assessments : An International Journal, 9 (6), 1431-1443.

Cormier S.M., Messer J.J. (2004). Opportunities and challenges in surface water quality monitoring. In: Wiersma GB, editor. Environmental monitoring. CRC Press, Boca Raton (FL), USA, 217-238.

Cormier S.M., Paul J.F., Spehar R.L., Shaw-Allen P., Berry W.J., Suter, G.W.II. (2008). Using field data and weight of evidence to develop water quality criteria. Integrated Environmental Assessment and Management, 4,490–504.

Cormier S.M., Suter G.W. II. (2008). A framework for fully integrating environmental assessments. Journal of Environmental Management, 42, 543–556.

Cormier, S.M., Suter, G.W.II, Norton, S.B. (2010). Causal Characteristics for Ecoepidemiology. HUman and Ecological risk assessment : An International Journal, 16(1), 53-73.

Cormier, S.M., Suter, G.W. II. (2013). A method for assessing causation of field exposure–response relationships. Environmental Toxicology and Chemistry, 32(2), 272–276.

Cormier, S.M., Suter G.W. II, Zheng L., Pond G.P. (2013). Assessing causation of the extirpation of stream macroinvertebrates by a mixture of ions. Environmental Toxicology and Chemistry, 32, 277–287.

Costanza, R., Mageau, M. (1999). What is a healthy ecosystem?. Aquatic Ecology, 33(1), 105-115.

CPCB (Central Pollution Control Board) (2008a). Guidelines for Water Quality Management. Central Pollution Control Board. 'Parivesh Bhawan', East Arjun Nagar, Dehli. http://www.cpcb.nic.in/upload/NewItems/NewItem_97_guidelinesofwaterqualitymanagement.pdf [Accessed 02 February 2016]

CPCB (Central Pollution Control Board) (2008b). Guidelines for Water Quality Monitoring. MINARS/27/2007-08. Central Pollution Control Board. 'Parivesh Bhawan', East Arjun Nagar, Dehli http://cpcb.nic.in/upload/NewItems/NewItem_116_Guidelinesof%20waterqualitymonitoring_31.07.08.pdf [Accessed 02 February 2016]

CSIRO (Commonwealth Scientific and Industrial Research Organisation) (1996). Review of riverine impacts. Porgera Joint Venture.

Culver, D.C., Holsinger, J.R. (1992). How many species of troglobites are there?. Bulletin of the National Speleological Society, 54, 59-80.

Cunha, D.G.F., Grull, D., Damato, M., Blum, J.R.C, Eiger, S., Lutti, J.E.I, Mancuso, P.C.S. (2011).

Contiguous urban rivers should not be necessarily submitted to the same management plan: the case of Tietê and Pinheiros Rivers (São Paulo – Brazil). Annals of the Brazilian Academy of Sciences, 83(4), 1465-1479.

Cuttelod, A., Seddon, M., Neubert, E. (2011). European Red List of non-marine molluscs. Publication Office of the European Union, Luxembourg, 60 pp.

Dallas, H.F., Day, J.A. (1993). The effect of water quality variables on riverine ecosystems: a review. Water Research Commission Report No. TT 61/93. Water Research Commission, Pretoria. 240 pp.

Dallas, H. F., Rivers-Moore, N. A. (2012). Critical thermal maxima of aquatic macroinvertebrates: towards identifying bioindicators of thermal alteration. Hydrobiologia, 679(1), 61-76.

Darwall, W.R.T., Smith, K.G., Allen, D., Seddon, M.B., Reid, G.M., Clausnitzer, V., Kalkman, V.J. (2009). Freshwater biodiversity: a hidden resource under threat. In: Vié, J.-C., Hilton-Taylor, C., Stuart, S.N. (Eds.). Wildlife in a Changing World. IUCN, Gland, pp. 43–54.

Darwall, W., Smith, K., Allen, D., Holland, R., Harrison, I., Brooks, E. (eds.) (2011a). The diversity of life in African freshwaters: underwater, under threat. IUCN, Cambridge and Gland.

Darwall, W.R.T, Holland, R.A., Smith, K.G., Allen, D., Brooks, E.G.E., Katarya, V., Pollock, C.M., Shi, Y., Clausnitzer, V., Cumberlidge, N., Cuttelod, A., Dijkstra, K.-D.B., Diop, M.D., García, N., Seddon, M.B., Skelton, P.H., Snoeks, J., Tweddle,D., Vié, J.-C. (2011b). Implications of bias in conservation research and investment for freshwater species. Conservation Letters, 4, 474–482.

Darwall, W. (2013). Freshwater Biodiversity and Protected Areas in Africa: a Gap Analysis. [Accessed through the Global Freshwater Biodiversity Atlas (atlas.freshwaterbiodiversity.eu) 25 February 2016]

Dávid, L., Telegdi, I. (1986). The Influence of Watershed Development on Long-Term Eutrophication of Lake Balaton. In: Somlyódy, L., van Straten, G., (eds.) (1986): Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

David, G.S. , Carvalho, E.D., Lemos, D., Silveira, A.N., Dall'Aglio-Sobrinho,M. (2015). Ecological carrying capacity for intensive tilapia (Oreochromis niloticus) cage aquaculture in a large hydroelectrical reservoir in Southeastern Brazil. Aquaculture Engineering, 66, 30 -40.

Davies, S.P., Jackson, S.K. (2006). The Biological Condition Gradient: A conceptual model for interpreting detrimental change in aquatic ecosystems. Ecological Applications, 16(4), 1251-1266.

Davis, R., Hirij, R. (eds.)(2003). Lake Management. Water Resources and Environment Technical Note G.2. World Bank , Washington, D.C,.

Davy-Bowker, J., Clarke, R.T., Johnson, R.K., Kokes, J., Murphy, J.F., Zahrádková, S. (2006) A comparison of the European Water Framework Directive physical typology and RIVPACS-type models as alternative methods of establishing reference conditions for benthic macroinvertebrates. Hydrobiologia 566, 91– 105.

Degefu, F., Mengistu, S., Schagerl, M. (2011). Influence of fish cage farming on water quality and plankton in fish ponds: A case study in the Rift Valley and North Shoa reservoirs, Ethiopia. Aquaculture, 316, 129–135.

De Groot, R., Brander, L., Van der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Crossman, N., Hein, L., Hussain, S., (2012). Global estimates of the value of ecosystems and their services in monetary units, Ecosystem Services, 1(1), 50-61.

Dennison, W.C., Lookingbill, T.R., Carruthers, T.J.B., Hawkey, J.M., Carter, S.L. (2007). An eye-opening approach to developing and communicating integrated environmental assessments. Frontiers in Ecology and the Environment,, 5: 307–315

Denys, L. (2004). Relation of abundance-weighted averages of diatom indicator values to measured environmental conditions in standing freshwaters. Ecological Indicators, 4, 255-275.

Devkota, B., Imberger, J. (2012). Upper and Middle Tietê River Basin dam-hydraulic system, travel time and temperature modeling. Journal of Hydrology 475 (2012) 12-25.

Dewulf, A., Craps, M., Bouwen, R., Taillieu, T., Pahl-Wostl, C. (2005). Integrated management of natural resources: Dealing with ambiguous issues, multiple actors and diverging frames. Water Science and Technology, 52(6), 115–124.

DeZuane, J. (1997). Handbook of drinking water quality. John Wiley & Sons, New York.

Dieperink, C. (1998). From open sewer to salmon run: lessons from the Rhine water quality regime. Water Policy, 1(5), 15.

Dinar, S., Katz, D., De Stefano, L., Blankespoor, B. (2015). Climate change, conflict, and cooperation: Global analysis of the effectiveness of international river treaties in addressing water variability. Political Geography, 45, 55-66.

Dinsmore, W.P., Scrimgeour, G.J., Prepas, E.E. (1999). Empirical relationships between profundal macroinvertebrate biomass and environmental variables in boreal lakes of Alberta, Canada. Freshwater Biology, 41, 91-100.

Di Toro, D.M., Allen, H.E., Bergman, H.L., Meyer, J.S., Paquin, R., Santore R.C. (2001). Biotic ligand model of the acute toxicity of metals. 1. Technical basis. Environmental Toxicolology and Chemistry, 20, 2383–2396.

Doberstein, C.P., Karr, J.R., Conquest, L.L. (2000). The effect of fixed-count subsampling on macroinvertebrate biomonitoring in small streams. Freshwater Biology, 44, 355-371.

Dodkins, I., Rippey, B. (2007). Objectivity and Confidence in Ecological Assessments of Rivers and Lakes. In: Heikkinen, E. (ed.). Focus on Water Resource Research. Nova Publishers, 149-183.

Doledec, S., Statzner, B. (2008). Invertebrate traits for the biomonitoring of large European rivers: An assessment of specific types of human impact. Freshwater Biology, 53(3), 617-634.

Döll, P., Bunn, S.E. (2014). Impact of Climate Change on Freshwater Ecosystems due to Altered River Flow Regimes. Box CC-RF. In: Field, C.B., Barros, V.R., Dokken, D.H., Mach, K.J., Mastrandrea, M.D.,Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., White, L.L. (eds.). Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge and New York .

Donohue, I., Irvine, K. (2007). Quantifying variability within water samples: The need for adequate subsampling. Water Research, 42, 476 – 482.

Donohue, I., Jackson, A.L., Pusch, M.T., Irvine, K., (2009). Nutrient enrichment homogenizes lake benthic assemblages at local and regional scales. Ecology, 90, 3470-3477.

Dore, J., Lebel, L., Molle, F. (2012). A framework for analyzing transboundary water governance complexes, illustrated in the Mekong Region. Journal of Hydrology, 466-467, 23-36.

Downes, B.J., Barmuta, L.A., Fairweather, P.G., Faith, D.P., Keough, M.J., Lake, P.S., Mapstone, B.D., Quinn, G.P. (2002). Monitoring ecological impacts: concepts and practice in flowing water. Cambridge University Press, New York.

Downing, J.A., Prairie, Y.T. , Cole, J.J., Duarte, C.M., Tranvik, L.J., Striegl R.G., McDowell, W.H., Kortelainen, P., Caraco, N.F., Melack, J.M., Middelburg, J.J. (2006), The global abundance and size distribution of lakes, ponds, and impoundments, Limnology and Oceanography, 51(5), 2388-2397.

Downing, J.A. (2009). Plenary lecture global limnology: Up–scaling aquatic services and processes to planet Earth. Verhandlungen des Internationalen Verein Limnologie, 30, 1149-1166.

DSEWPC (Department of Sustainability, Environment, Water, Population and Communities) (2009). Compliance and Enforcement Policy, Government Australia https://www.environment.gov.au/about-us/accountability-reporting/compliance-enforcement-policy [Accessed 02 February 2016]

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z. I., Knowler, D. J., Lévêque, C., Naiman, R.J., Prieur-Richard, A-H., Soto, D., Stiassny, M.L.J., Sullivan, C.A. (2006). Freshwater biodiversity: Importance, threats, status and conservation challenges. Biological Reviews, 81(2), 163-182.

Dugan, P., Sugunan, V. V., Welcomme, R. L., Bene C., Brummett, R.E., Beveridge, M.C.M., Kofi, A., Amerasinghe, U., Arthington, A., Blixt, M., Chimatiro, S., Katiha, P., King, J., Kolding,J., Khoa, S.N., Turpie, J. (2007). Inland fisheries and aquaculture. In: Molden, D. (ed.). Water for food, water for life: A Comprehensive Assessment of Water Management in Agriculture. Earthscan and International Water Management Institute, 459-483.

DWAF (Department of Water Affairs and Forestry) (1996a). Water Quality Guidelines, Volume 7, Aquatic Ecosystems. Department of Water Affairs and Forestry, Republic of South Africa. https://www.capetown.gov.za/en/CSRM/Documents/Aquatic_Ecosystems_Guidelines.pdf [Accessed 02 February 2016]

DWAF (Department of Water Affairs and Forestry) (1996b). Water Quality Guidelines, Volume 8, Field Guidelines. Department of Water Affairs and Forestry, Republic of South Africa. http://www.iwa-network.org/filemanager-uploads/WQ_Compendium/Database/Future_analysis/078.pdf [Accessed 02 February 2016]

DWAF (Department of Water Affairs and Forestry) (1998). National Water Act. https://www.dwa.gov.za/Documents/Legislature/nw_act/NWA.pdf [Accessed 02 February 2016]

DWAF (Department of Water Affairs and Forestry). (2001). Crocodile, Sabie/Sand & Olifants River Systems. State of Rivers Report, March 2001. http://www.dwaf.gov.za/iwqs/rhp/state_of_rivers/crocsabieolif_01_toc.html [Accessed 19 February 2016].

DWAF (Department of Water Affairs and Forestry) (2003). Resource directed measures. Module 1: introductory module. Pretoria, South Africa.

DWAF (Department of Water Affairs and Forestry). (2008). River Health Programme – Mthatha River. South African National Aquatic Ecological Health Monitoring Programme. https://www.dwaf.gov.za/iwqs/rhp/state_of_rivers/posters/Mthatha-6-e.pdf [Accessed 19 February 2016].

DWA (Department of Water Affairs) (2011). Directorate Water Resource Planning Systems: Water Quality Planning. Resource Directed Management of Water Quality. Planning Level Review of Water Quality in South Africa. Sub-series No.WQP 2.0. Pretoria, South Africa.

DWA (Department of Water Affairs) (2013). Strategic overview of the water sector in South Africa. DWA, Pretoria, South Africa. 73 pp.

DWA (Department of Water Affairs) (2014). Resource Directed Measures Compliance. 2014. Determination of Resource Quality Objectives in the Olifants Water Management Area (WMA4) – WP10536. Draft Resource Quality Objectives Report (version 1): Stakeholder Engagement Process. March 2014. Prepared by the Institute of Natural Resources (INR) NPC. Pietermaritzburg, South Africa.

DWS (Department of Water and Sanitation) (2014). Determination of Resource Quality Objectives in the Olifants Water Management Area (WMA4): Resource quality objectives and numerical limits report. Report No.: RDM/WMA04/00/CON/RQO/0613. Directorate: Resource Directed Measures: Compliance. Study No.: WP10536. Prepared by the Institute of Natural Resources (INR) NPC. INR Technical Report. No.: INR 492/14.6. Pietermaritzburg, South Africa.

Dyson, M., Bergkamp, G. Scanlon, J. (2003). Flow: the essentials of environmental flows. IUCN, Gland and Cambridge. www.iucn.org/about/work/programmes/water/resources/toolkits/?2186 [Accessed 02 February 2016]

EC (European Commission) (2000a). Water Framework Directive (WFD) 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy, Official Journal of the European Communities L327, 1-72.

EC (European Commission) (2000b). Common Implementation Strategy for the Water Framework Directive (2000/60/EC) Guidance Document No. 27, Technical Guidance for Deriving Environmental Quality Standards. https://circabc.europa.eu/sd/a/0cc3581b-5f65-4b6f-91c6-433a1e947838/TGD-EQS%20CIS-WFD%2027%20EC%202011.pdf [Accessed 02 February 2016]

EC (European Commission) (2003). Guidance document No 10. Rivers and Lakes – Typology, Reference Conditions and Classification Systems. European Commission, Brussel.

EC (European Commission) (2008) Directive 2008/105/EC of the European Parliament and the Council of 16 December 2008 on environmental quality standards. http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:32008L0105 [Accessed 02 February 2016]

EC (European Commission) (2009a). Guidance document no 23, Guidance document on eutrophication assessment in the context of European policies. European Commission, Brussels.

EC (European Commission) (2009b). Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy. Consolidated version, June 2009.

EC (European Commission). (2011a). Common Implementation Strategy for the Water Framework Directice (2000/60/EC). Guidance Document no. 14: Guidance on the intercalibration process 2008-2011. https://circabc.europa.eu/sd/a/61fbcb5b-eb52-44fd-810a-63735d5e4775/IC_GUIDANCE_FINAL_16Dec2010.pdf [Accessed 22 February 2016]

EC (European Commission) (2011b). Guidance document No 27. Technical Guidance For Deriving Environmental Quality Standards. https://circabc.europa.eu/sd/a/0cc3581b-5f65-4b6f-91c6-433a1e947838/TGD-EQS%20CIS-WFD%2027%20EC%202011.pdf [Accessed 18 February 2016]

EC (European Commission) (2012). Report from the Commission to the European Parliament and the Council on the Implementation of the Water Framework Directive (2000/60/EC). River Basin Management Plans. European Commission, Brussel.

EC (European Commission) (2013a). Directive 2013/39/EU of the European Parliament and Council of 12 August 2013 amending Directives 2000/60/EC and 2008/105/EC as regards priority substances in field of water policy. Annex II 'I Environmental Quality Standards for Priority Substances and Certain Other Pollutants, Part A: Environmental Quality Standards. Official Journal of the European Union. Vol. 56: L 226/14-17 http://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=OJ:L:2013:226:FULL&from=EN [Accessed 02 February 2016]

EC (European Commission) (2013b). Mapping and Assessment of Ecosystems and their Services. An analytical framework for ecosystem assessment under Action 5 of the EU Biodiversity Strategy.

EC (European Commission) (2013c). "Introduction to the new EU Water Framework Directive." from http://ec.europa.eu/environment/water/water-framework/info/intro_en.htm [Accessed 02 February 2016]

EC (European Commission). (2015). Procedure to fit new or updated classification methods to the results of a completed intercalibration exercise. Guidance Document no. 30. https://circabc.europa.eu/sd/a/5aee6446-276c-4440-a7de-0d4dec41ed4b/IC_manual_2015_to%20be%20published.pdf [Accessed 18 February 2016]

ECB (European Central Bank) (2003). Technical Guidance Document (TGD) on risk assessment of chemical substances following European Regulations and Directives, 2nd Edition.

EEA (European Environmental Agency) (2010). Scaling up ecosystem benefits. A contribution to the Economics of Ecosystems and Biodiversity (TEEB) study. Report No. 4. EEA, Copenhagen. http://dx.doi.org/10.2800/41295 [Accessed 02 February 2016]

Ellis, J. (2006). Uncertainty estimation for monitoring results by the WFD biological classification tools. Science Report to the UK Environment Agency.

Elosegi, A., Sabater, S. (2013). Effects of hydromorphological impacts on river ecosystem functioning: a review and suggestions for assessing ecological impacts. Hydrobiologia, 712, 129-143.

Emerson. K, Lund, R.E., Thurston R.V., Russo, R.C. (1975). Aqueous ammonia equilibrium calculations: effect of PH and temperature. Effect of pH and Temperature. Journal of the Fisheries Research Board of Canada, 32, 2379-2383.

Enderlein U.S., Enderlein, R.E., Williams, W.P. (1997). Water Quality Requirements. In: R. Helmer and I. Hespanhol, Water Pollution Control, UNEP/WHO, E & FN SPON, London.

Engel, S., Schäfer, A. (2013). Ecosystem services — a useful concept for addressing water challenges?. Current Opinion in Environmental Sustainability, 5, 696-707.

Entz, G., Sebestyén, 0. (1942). The life of Lake Balaton. Magyar Term. Tud. Társ. Budapest. (In Hungarian).

Entz, G., Sebestyén, 0. (1946). The life of Lake Balaton. Magyar Biol. Kut. Int. Munk. 16, 179-411. (In German).

Environment Canada (2013). Federal Environmental Quality Guidelines. Alcohol Ethoxylates. http://www.ec.gc.ca/ese-ees/164786DB-7B5B-47DF-93FA888C99E3D612/FEQG_Alcohol_Ethoxylates_EN.pdf [Accessed 02 February 2016]

Erba, S., Furse, M.T., Balestrini, R., Christodoulides, A., Ofenböck, T., van de Bund, W., Wasson, J-G. , Buffagni, A. (2009). The validation of common European class boundaries for river benthic macroinvertebrates to facilitate the intercalibration process of the Water Framework Directive. Hydrobiologia, 633, 17–31.

Ericson, J.P., Vörösmarty, C.J., Dingman, S.L., Ward, L.G., Meybeck, M. (2006). Effective sea-level rise in deltas. Sources of change and human-dimension implications. Global Planet Change , 50,63-82.

ESAWADI (Ecosystem Services Approach for Water Framework Directive Implementation) (2011). Utilizing the Ecosystem Services Approach for the Water Framework Directive Implementation. Framework of Analysis.

ESAWADI (Ecosystem Services Approach for Water Framework Directive Implementation) (2013). Synthesis Report. http://www.iwrm-net.eu/sites/default/files/ESAWADI%20SCP%20final%20synthesis%20july2013.pdf [Accessed 02 February 2016]

Evans-White, M. A., Haggard, B. E., Scott, J. T. (2013). A review of stream nutrient criteria development in the United States. Journal of Environmental Quality, 42(4), 1002-1014.

Fairchild, G.W., Faulds, A.M., Matta, J.F. (2000). Beetle assemblages in ponds: effects of habitat and site age. Freshwater Biology, 44, 523-534.

Fairweather, P.G. (1991). Statistical power and design requirements for environmental monitoring. Australian Journal of Marine and Freshwater Research, 42 (5), 555 -567.

Feeley K.J., Silman M.R. (2011). The data void in modeling current and future distributions of tropical species. Global Change Biology, 17, 626-630.

Felföldi, L. (1974). Biological water quality classification. VIZDOK, Budapest. (In Hungarian).

Finlayson, C.M., D'Cruz, R. (2005). Inland Water Systems. In: Hassan, H., Scholes, R., Ash, N. (eds.). Ecosystems and human well-being: current state and trends: findings of the Conditions and Trends Working Group. Millennium Ecosystem Assessment. Island Press, Washington D.C.

Finn M., Jackson S. (2011). Protecting indigenous values in water management: a challenge for conventional environmental flow assessments. Ecosystems, 1232-1248.

Fitch, K, Kemker, C. (2014). “Algae, Phytoplankton and Chlorophyll”. Fundamentals of Environmental Measurements. Fondriest Environmental, Inc. 22 Oct. 2014.

FGN (Federal Gouvernment of Nigeria) (1988). Federal Environmental Protection Agency Act (FEPAAct,1988). http://www.placng.org/new/laws/F10.pdf [Accessed 02 February 2016]

FGN (Federal Government of Nigeria) (1991). Interim Guidelines and Standards for Industrial Effluent, Gaseous Emissions and Noise Limitations, LEX-FAOC018380, http://faolex.fao.org/ [Accessed 02 February 2016]

Fjellheim, A., Raddum, G.G. (1990). Acid precipitation: biological monitoring of streams and lakes. Science of the Total Environment, 96, 57-66.

Flotemersch, J.E., Leibowitz,S.G., Hill, R.A., Stoddard,J.L., Thoms, M.C., Tharme, R.E. (2015). A Watershed Integrity Definition and Assessment Approach to Support Strategic Management of Watersheds. River Research and Applications.

Fluet-Chouinard, E., Lehner, B., Rebelo, Papa, F., L.-M., Hamilton, S. (2015). Development of a global inundation map at high spatial resolution from topographic downscaling of coarse-scale remote sensing data. Remote Sensing of Environment, 158, 348-361.

Freshwater Information Plattform (2006). http://www.freshwaterplatform.eu/ [Accessed 16 February 2016]

Focardi, S., Corsi, I., Mazzuoli, S., Vignoli, L., Loiselle, S., Focardi, S. (2006). Integrating remote sensing approach with pollution monitoring tools for aquatic ecosystem risk assessment and management: A case study of Lake Victoria (Uganda). Environmental Monitoring and Assessment, 122, 275-287.

Frey, D. (1975). Biological integrity of water: an historical perspective. In: Ballentine, R.K., Guarraia, L. J. (eds.). The Integrity of Water. US EPA, Washington, DC, 127-139.

Friberg, N., Bonda, N., Bradley, D.C., Dunbar, M.J., Edwards, F.K., Grey, J., Hayes, R.B., Hildrew, A.G., Lamouroux, N., Trimmer, M., Woodward, G. (2011). Bio monitoring of human impacts in freshwater ecosystems: the good, the bad and the ugly. Advances in Ecological Research, 44, 1-68.

Furse, M. T., Moss, D. (1984). The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. Freshwater Biology, 14, 257-280.

Furse, M. T., Clarke, R.T., Winder, J.M., Symes, K. L., Blackburn, J.H., Grieve, N.J., Gunn, R.J.M. (1995). Biological Assessment methods: Controlling the Quality ofBiological Data. Package 1: the Variability of Data Used for Assessing the Biological Condition of Rivers. R & D Note 412. National Rivers Authority, Bristol.

Furusjö, E. (2006). Applicability of QSAR models (Quantitative Structure Activity Relationships) for assessing impact of chemical compounds. In: Solimini, A.G., Cardosa, A. C., Heiskanen, A.S. (eds.). Indicators and methods for the ecological status assessment under the Water Framework Directive. European Commission, Joint Research Centre, Institute for Environment and Sustainability, 167-192.

FUSP (Fundação de Apoio à Universidade de São Paulo) (2002). Plano da Bacia Hidrográfica do Alto Tietê. Sumário Executivo. Fundação de Apoio à Universidade de São Paulo.

FUSP (Fundação de Apoio à Universidade de São Paulo) (2009). Plano da Bacia Hidrográfica do Alto Tietê. Fundação de Apoio à Universidade de São. Paulo.

Fuster, R., Escobar, C., Lillo, G., Gonzalez, M., de la Fuente, A., Pottigiesser, T. (2012). Water bodies typology system: A Chilean case of scientific stakeholders and policy maker’s dialogue. Lakes, reservoirs and ponds, 6(2), 93-107.

Garg, J., Garg, H.K. (2002). Nutrient loading and its consequences in a lake ecosystem. Tropical Ecology, 43(2), 355-358.

Garrido Pérez, A., Cuevas, M.L., Cotler, H., González, D.I., Tharme, R.E. (2010). Evaluación del grado de alteración ecohidrológica de los ríos y corrientes superficiales de México. Investigación Ambiental. Ciencia y Política Pública, 2(1), 25-46.

Garrido, A., Cuevas, M.L., Cotler, H., González, D.I., Tharme, R. (2010b). El estado de alteración ecohidrológica de los ríos de México. Pp 108-111. In: Cotler Ávanos, H. (Coord.). Las cuencas hidrográficas de México: diagnóstico y priorización. Instituto Nacional de Ecología-Fundación Gonzalo Río Arronte I.A.P., México City, México. Pluralia Ediciones e Impresiones, México. 232 pp.

Gabriels, W., Lock, K., De Pauw, N., Goethals, P.L.M. (2010). Multimetric Macroinvertebrate Index Flanders (MMIF) for biological assessment of rivers and lakes in Flanders (Belgium). Limnologica, 40, 199-207.

Gerritsen, J, J. Burton, M. T. Barbour. 2000. A Stream Condition Index for West Virginia Wadeable Streams. http://www.dep.wv.gov/WWE/watershed/bio_fish/Documents/WVSCI.pdf [Accessed 02 February 2016]

Gerritsen, J., L. Zheng, J. Burton, C. Boschen, S. Wilkes, J. Ludwig, and S. Cormier (2010). Inferring Causes of Biological Impairment in the Clear Fork Watershed, West Virginia. US EPA, Office of Research and Development, National Center for Environmental Assessment, Cincinnati, OH. EPA/600/R-08/146.

Gessner, M.O., Chauvet, E. (2002). A case for using litter breakdown to assess functional stream integrity. Ecological Applications, 12, 498–510.

Gibson C.E., Wang, G., Foy, R.H., Lennox, S.D. (2001). The importance of catchment and lake processes in the phosphorus budget of a large lake. Chemosphere, 42, 215-220.

Gilvear, D.J., Davids, C., Tyler, A.N. (2004). The use of remotely sensed data to detect channel hydromorphology; River Tummel. Scotland River research and applications, 207, 795-811.

Gondwe, M. J.S., Guildford, S.J., Hecky; R.E. (2011). Carbon, nitrogen and phosphorus loadings from tilapia fish cages in Lake Malawi and factors influencing their magnitude. Journal of Great Lakes Research. 37, 93-101.

Goulder, L.H., Kennedy, D. (1997). Valuing ecosystem services: philosophical bases and empirical methods. In: Daily G, (ed.). Nature's services. Island Press, Washington DC, 23–48.

GovBC (Government of British Columbia, Ministry of Environment) (2001). Water Quality Guidelines for Temperature. http://www.env.gov.bc.ca/wat/wq/BCguidelines/temptech/temperature.html [Accessed 11 March 2016]

Green, O.O., Garmestani, A.S., Van Rijswick, H.F.M.W., Keessen, A.M. (2013). EU water governance: striking the right balance between regulatory flexibility and enforcement? Ecology and Society, 18(2), 10.

Grill, G., Lehner, B., Lumsdon, A.E., MacDonald, G.K., Zarfl, C., Reidy Liermann, C. (2015). An index-based framework for assessing patterns and trends in river fragmentation and flow regulation by global dams at multiple scales. Environmental Research Letters, 10, 015001.

Groffman, P., Baron, J., Blett, T., Gold, A., Goodman, I., Gunderson, L., Levinson, B., Palmer, M., Paerl, H., Peterson, G., LeRoy Poff, N., Rejeski, D., Reynolds, J., Turner, M., Weathers, K., Wiens, J. (2006). Ecological thresholds: the key to successful environmental management or an important concept with no practical application? Ecosystems, 9(1), 1–13.

GSDI (Global Spatial Data Infrastructures Association) (2013). The SDI Cookbook. http://www.gsdidocs.org/GSDIWiki/index.php/Main_Page.

Guilpart, A., Roussel, J. M., Aubin, J., Caquet, T., Marle, M., Le Bris, H. (2012). The use of benthic invertebrate community and water quality analyses to assess ecological consequences of fish farm effluents in rivers. Ecological Indicators, 23, 356-365.

Guo, L., Li., Z. (2003). Effects of nitrogen and phosphorus from fish cage-culture on the communities of a shallow lake in middle Yangtze River basin of China. Aquaculture, 226, 201 – 212.

GWP (Global Water Partnership) (2014). River basin organizations (B1.04). http://www.gwp.org/en/ToolBox/TOOLS/Institutional-Roles/Creating-an-Organisational-Framework/River-basin-organisations/ [Accessed 02 February 2016]

Hackenson, L. (2005). Changes to lake ecosystem structure resulting from fish cage farm emissions, Lakes and Reservoir Management, 10, 71-80.

Hajibabaei, M., Shokralla, S., Zhou, X., Singer G.A.C., Baird D.J. (2011). Environmental Barcoding: A Next-Generation Sequencing Approach for Bio monitoring Applications Using River Benthos. PLOS ONE, 6(4), 1-7.

Håkanson, L. (1999). On the principles and factors determining the predictive success of ecosystem models, with a focus on lake eutrophication models. Ecological Modelling, 121, 139-160.

Hallare, A.V., Factor, P.A., Santos, E.K., Hollert, H. (2009). Assessing the Impact of Fish Cage Culture on Taal Lake (Philippines) Water and Sediment Quality Using the Zebrafish Embryo Assay. Philippine Journal of Science, 138 (1), 91-104.

Hallare, A.V., Lasafin, K.J.A., Magallanes, J.R. (2011). Shift in Phytoplankton Community Structure in a Tropical Marine Reserve Before and After a Major oil Spill Event. International Journal of Environmental Research, 5(3), 651-660.

Halse, S.A., Cale, D.J., Jasinka, E.J., Shiel, R.J. (2002). Monitoring change in aquatic invertebrate biodiversity: Sample size, faunal elements and analytical methods. Aquatic Ecology, 36, 395-410.

Hämäläinen, H., Luotonen, H., Koskenniemi, E., Liljaniemi, P. (2003). Inter-annual variation in macroinvertebrate communities in a shallow forest lake in eastern Finland during 1990-2001. Hydrobiologia, 506/509, 389-397.

Harremoës, G., Gee, D., MacGarvin, M., Stirling, D., Keys, J., Wynne, B., Guedes Vaz, S. (2002). Late Lessons from Early Warnings: the Precautionary Principle 1896–2000. Environmental Issue Report, 22. European Environment Agency.

Harremoes, P. (2003). The need to account for uncertainty in public decision making related to technological change. Integrated Assessment, 4, 18–25.

Harris, N.M., Gurnell, A.M., Hannah, D.M., Petts, G.E. (2000). Classification of river regimes: a context for hydroecology. Hydrological Processes, 14, 2831–2848.

Hashimoto, T., Stedinger, J.R., Loucks, D.P. (1982). Reliablity, resiliency and vulnerability criteria for water resources system performance evaluation. Water Resources Research, 18(1), 14-20.

Haury, J., Peltre, M.-C., Trémolières, M., Barbe, J., Thiébaut, G., Bernez, I., Daniel, H., Chaternet, P., Haan-Archipof, G., Muller, S., Dutartre, A., Laplace-Treyture, C., Cazaubon, A., Lambert-Servien, E. (2006). A new method to assess water trophy and organic pollution - the Macrophyte Biological Index for Rivers (IBMR): it's application to different types of river and pollution. Hydrobiologia, 570, 153-158.

Hawkins, C.P., Norris, R.H., Gerritsen, J., Hughes, R.M., Jackson, S.K., Johnson, R.K., Stevenson, R.J. (2000). Evaluation of landscape classifications for biological assessment of freshwater ecosystems: synthesis and recommendations. Journal of the North American Benthological Society, 19, 475-486.

Hawkins C.P. (2006) Quantifying biological integrity by taxonomic completeness: its utility in regional and global assessments. Ecological Applications, 16, 1277– 1294.

Hecky, R.E., Mugidde, R., Ramlal, P.S., Talbot, M.R., Kling, G.W. (2010). Multiple stressors cause rapid ecosystem change in Lake Victoria. Freshwater Biology, 55(s1), 19-42.

Heino, J. (2000). Lentic macroinvertebrate assemblage structure along gradients in spatial heterogeneity, habitat size and water chemistry. Hydrobiologia, 418, 229-242.

Heino, J., Virkkala, R., Toivonen, H. (2009). Climate change and freshwater biodiversity: Detected patterns, future trends and adaptations in northern regions. Biological Reviews, 84(1), 39-54.

Hellawell, J.M. (1986). Biological Indicators of Freshwater Pollution and Environmental Management. Elsevier Applied Science Publishers, London.

Helmer, R., Hespanhol, I (eds.) (1997). Water pollution control - a guide to the use of water quality management principles. St Edmundsbury Press, Suffolk.

Henny, C. (2009). Dynamics of Biogeochemistry of Sulfur in Lake Maninjau. LIMNOTEK VII(2), p. 34-35.

Henny, C., Nomosatryo, S. (2012). Nutrient Dynamics in Lake Maninjau: Implication of fish cage fishery. The Tenth International Symposium on South East Asia Water Environment. November 7 – 11. Hanoi Vietnam, p. 170-177.

Henny, C. (2014). Ecotechnology and ecosystem based management tools as alternative solutions to reduce risks of ecosystem disaster and fish kill in Lake Maninjau, Proceeding of National Seminar of Limnology, 17 October, Cibinong, 593-607.

Hering, D., Feld, C.K., Moog, O., Ofenbök, T. (2006). Cook book for the development of a Multimetric Index for biological condition of aquatic ecosystems: experiences from the European AQEM and STAR projects and related initiatives. Hydrobiologia, 566, 311-324.

Hering, D., Borja, A., Carstensen, J., Carvalho, L., Elliott, M., Feld, C.K., Heiskanen, A-S., Johnson, R.K., Moe, J., Pont, D., LycheSolheim, A., van de Bund, W. (2010). The European Water Framework Directive at the age of 10: A critical review of the achievements with recommendations for the future. Science of the Total Environment, 408, 407-4019.

Hering, D., Birk, S., Solheim, A.L., Moe, J., Carvalho, L., Borja, A., Hendriksen, P., Krause-Jensen, D., Lauridsen, T., Sondergaard, M., Pont, D., Johnson, R., Kolada, A., Porst, G., Marba, N., Noges, P., Ott, I., Marques, J. C., Irvine, K., Basset, A. (2012). Guidelines for indicator development. Water bodies in Europe: Integrative Systems to Assess Ecological status and Recovery (WISER). http://www.wiser.eu/download/D2.2-2.pdf [Accessed 19 February 2016]

Herlihy, A.T., Paulsen, S.G., Van Sickle, J., Stoddard, J.L., Hawkins, C.P., Yuan, L.L. (2008). Striving for consistency in a national assessment: the challenges of applying a reference-condition approach at a continental scale. Journal of the North American Benthological Society, 27, 860-877.

Herodek, S. (1986). Phytoplankton Changes during Eutrophication and P and N Metabolism. In: Somlyódy, L., van Straten, G. (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Hickey C.W. , Pyle, E. (2001). Derivation of water quality guideline values for heavy metals using a risk-based methodology: an approach for New Zealand. Australasian Journal of Ecotoxicology, 7, 137-156.

Higgins, J.V., Bryer, M.T., Khoury, M.L., Fitzhugh T.W. (2005). A freshwater classification approach for biodiversity conservation planning. Conservation Biology, 19, 432-445.

Hill, R.A., Weber, M.H., Leibowitz, S.G., Olsen, A.R., Thornbrugh, D.J. (2016). The Stream-Catchment (StreamCat) Dataset: A database of watershed metrics for the conterminous USA. Journal of the American Water Resources Association, 52(1), 120-128.

Hinck, J.E., Blazer, V.S., Schmitt, C.J., Papoulias, D.M., Tillitt, D.E. (2009). Widespread occurrence of intersex in black basses (Micropterus spp.) from US rivers, 1995–2004. Aquatic Toxicology, 95(1), 60-70.

Holland, R.A., Garcia, N., Brooks, E.G.E., Juffe, D., with contributions from: Howard, G. (Section 8.2.3); Foster, M.N. (Section 8.3.4); Harrison, I., Upgren, A., Alonso, L.E. (Section 8.5); Matthews, J.H., Wickel, A.J., Freeman, S., Thieme, M.L. (2011). Chapter 8 - Synthesis for all taxa. In: Darwall, W.R.T., Smith, K.G., Allen D.J., Holland, R., Harrison, I., Brooks, E. (eds). The diversity of life in African freshwaters: under water, under threat. An analysis of the status and distribution of freshwater species throughout mainland Africa. IUCN, Gland and Cambridge.

Holland, R.A., Darwall, W.R.T., Smith, K.G. (2012). Conservation priorities for freshwater biodiversity: The Key Biodiversity Area approach refined and tested for continental Africa. Biological Conservation, 148, 167–179.

Holling, C.S. (1973). Resilience and stability of ecological systems. Annual Review of Ecology and Systematics, 4, 1-23.

Howarth, W. (2006). The progress towards ecological quality standards. Journal of Environmental Law, 18:3-35

Hughes, D.A., Hannart P. (2003). A desktop model used to provide an initial estimate of the ecological instream flow requirements of rivers in South Africa. Journal of Hydrolofy. 270, 167-181.

Hughes D.A. (ed.) (2004). SPATSIM, an integrating framework for ecological Reserve determination and implementation: incorporating water quality and quantity components for rivers. Water Research Commission, Pretoria, South Africa.

Hunsaker, C.T., Carpenter, D.E. (1990). Ecological Indicators for the Environmental Monitoring and Assessment Program. EPA 600/3-90/060. The US Environmental Protection Agency. Office of Research and Development. Research Triangle Park, N.C.

Huntley, B.J. (1996). South African experience regarding alien species: impacts and controls. In: Sandlund, O.T., Schei, P.J., Viken, A. (eds.). Proceedings of Norway/UN Conference on Alien Species. The Norwegian Ministry of Environment, Trandheim, 182–188.

HW (Healthy Waterways). (2016). The Ecosystem Health Monitoring Program. http://www.ehmp.org/ehmphome.aspx [Accessed 24 February 2016].

ICPDR (International Commission for the Protection of the Danube River basin) (2016). http://icpdr.org/main/ [Accessed 18 February 2016]

IBGE (Brazilian Institute of Geography and Statistics) (2010). Population Census 2010. http://www.ibge.gov.br/english/estatistica/populacao/censo2010/ [Accessed 24 February 2016]

IIED (International Institute for Environment and Development), WBCSD (World Business Council for Sustainable Development) (2002). Mining for the Future. Appendix I: Porgera Riverine Disposal Case Study. http://pubs.iied.org/pdfs/G00562.pdf [Accessed 09 March 2016]

International Water Centre (2016). Report Cards. http://www.watercentre.org/portfolio/rhef/project-resources/report-cards/reportcards [Accessed 24 February 2016].

Irvine, K. (2004). Classifying ecological status under the European Water Framework Directive: the need for monitoring to account for natural variability. Aquatic Conservation: Marine and Freshwater Systems, 14, 107-112.

Irvine, K. (2012). The tragedy of the threshold: Revising perceptions for aquatic conservation. Aquatic Conservation: Marine and Freshwater Ecosystems, 22, 705-711.

ISO (International Organization for Standadization). (2005). General requirements for the competence

of testing and calibration laboratories. ISO/IEC 17025. http://www.iso.org/iso/home/store/catalogue_tc/catalogue_detail.htm?csnumber=39883 [Accessed 18 February 2016]

ISO (International Organization for Standadization). (2012). Guidance on Project Management. ISO 21500:2012. http://www.iso.org/iso/catalogue_detail?csnumber=50003 [Accessed 18 February 2016]

ISTAT (Italian National Institute of Statistics), Constantino, C., Falcitelli, F., Femia, A., Tuolini, A. (2003). Conceptual framework DPSIR model. OECD Workshop Paris, May 14-16 2003.

Istvánovics, V., Clement, A., Somlyódy, L., Specziár, A., Tóth, L.-G., Padisák, J. (2007). Updating water quality targets for shallow Lake Balaton (Hungary), recovering from eutrophication. Hydrobiologia. 581, 305–318.

IUCN (International Union for Conservation of Nature) (2007). Identification and Gap Analysis of Key Biodiversity Areas – Targets for Comprehensive Protected Area Systems. World Commission on Protected Areas, Best Practice, Protected Area Guidelines Series No. 15, IUCN, Gland.

IUCN (International Union for Conservation of Nature) (2013). The IUCN Red List of threatened species 2013. http://www.iucnredlist.org [Accessed 02 February 2016]

Jackson, L. E., Kurtz, J.C., Fisher, W.S. (eds.) (2000). Evaluation Guidelines for Ecological Indicators. EPA/620/R-99/005. US EPA, Office of Research and Development, Wahington, D.C.

Jackson, J.B.C., Kirby, M.X., Berger, W.H., Bjorndal, K.A., Botsford, L.W., Bourque, B.J., Bradbury, R.H., Cooke, R., Erlandson, J., Estes, J.A., Hughes, T.P., Kidwell, S., Lange, C.B., Lenihan, H.S., Pandolfi, J.M., Peterson, C.H., Steneck, R.S., Tegner, M.J., Warner, R.R. (2001). Historical Overfishing and the Recent Collapse of Coastal Ecosystems. Science, New Series, 293(5530), 629-638.

Jackson, C. R. (2006). Wetland hydrology. In: Batzer, D. P., Sharitz, R.R. (eds.). Ecology of freshwater and estuarine wetlands. University of California Press, 43-81.

Jackson, L.E., Paulsen, S.G. (2009). Preface: Special issue: The Eighth Symposium of the Environmental Monitoring and Assessment Programme (EMAP)—research and partnerships for accountability. Environmental Monitoring and Assessment, 150, 1-2.

Jarvis, T., Wolf, A. (2010). Managing Water Negotiations and Conflicts in Concept and Practice. In A. Earle et al., eds. Transboundary Water Management: Principles and Practice. Earthscan, London and Washington, pp. 125–142.

Jaspers, F. G. (2003) Institutional Arrangements for Integrated River Basin Management. Water Policy 5, 77-90.

Jelks, H.L., Walsh, S.J., Burkhead, N.M., Contreras-Balderas, S., Díaz-Pardo, E., Hendrickson, D.A., Lyons, J., Mandrak, N.E., McCormick, F., Nelson, J.S., Platania, S.P., Porter, B.A., Renaud, C.B., Schmitter-Soto, J.J., Taylor, E.B., Warren, M.L. (2008). Conservation status of imperiled North American freshwater and diadromous fishes. Fisheries, 33, 372-407.

Jeppesen, E., Søndergaard, M., Jensen, J.P., Havens, K., Anneville, O., Carvalho, L., Coveney, M. F., Denke, R., Dokulil, M.T., Foy, B., Geradeaux, T., Hampton, S.E., Hilt, S., Kangur, K., Köhler, J., Lammens, E. H.H.R., Lauridsen, T.L., Manca, M., Miracle, M., Moss, B., Nõges, P., Persson, G., Phillips, G., Portielje, R., Romo, S., Schelske, C.L., Straile, D., Tatrai, I., Wille, E., Winder, M. (2005). Lake responses to reduced nutrient loading—an analysis of contemporary long-term data from 35 case studies. Freshwater Biology, 50, 1747–71.

Jobling, S., Nolan, M., Tyler, C. R., Brighty, G., Sumpter, J. P. (1998). Widespread sexual disruption in wild fish. Environmental Science & Technology, 32(17), 2498-2506.

Johnson, R.K. (1998). Spatiotemporal variability of temperate lake macroinvertebrate communities: detection of impact. Ecological Applications, 8, 61-70.

Johnson, K.J. (2000). Spatial congruence between ecoregions and littoral macroinvertebrate assemblages. Journal North American Benthological Society, 19, 475 – 486.

Johnson, R.K., Goedkoop, W., Sandin, L. (2004). Spatial scale and ecological relationships between the macroinvertebrate communities of stoney habitats of streams and lakes. Freshwater Biology, 49, 1179-1194.

Johnson, R.K., Hering D., Furse M.T., Clarke R.T. (2006). Detection of ecological change using multiple organism groups: metrics and uncertainty. Hydrobiologia, 566, 115-137.

Jones, I.J., Clarke, R.T., Blackburn, J.H., Gunn, R.J.M., Kneebone, N.T, Neale, M.W. (2007). Lake benthic macroinvertebrates II: Quantifying uncertainty in sampling methodology. Science Report: SC030294/SR2. U.K. Environmental Agency, Bristol.

Jordan, A., O'Riordan, T. (2004). The precautionary principle: a legal and policy history. In: Martuzzi, M., Tickner, J.A. (eds.). The precautionary principle: protecting public health, the environment and the future of our children. World Health Organization, 31-48.

Junk, W.J., Bayley, P.B., Sparks, R.E. (1989). The flood pulse concept in river-floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences, 106(1), 110-127.

Jyväsjärvi J., Tolonen K.T., Hämäläinen H. (2009). Natural variation of profundal macroinvertebrate communities in boreal lakes is related to lake morphometry: implications for bioassessment. Canadian Journal of Fisheries and Aquatic Science, 66, 589-601.

Karr, J.R. (1981). Assessment of biotic integrity using fish communities. Fisheries, 6(6),21-27.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I. J. (1986). Assessing biological integrity in running waters: a method and its rationale. 5, Illinois Natural History Survey.

Karr, J.R. (1995). Ecological integrity and ecological health are not the same. In: Schulze, P. (ed.). Engineering within ecological constraints. National Academy of Engineering, National Academy Press, 1-15.

Karr, J. R. (1999). Defining and measuring river health. Freshwater Biology, 41, 221–234.

Karr, J.R., Chu, E.W. (1999). Restoring Life in Running Waters. Better Biological Monitoring. Island Press, 207pp.Karr, J.R., Yoder C. O. (2004). Biological assessment and criteria improve total maximum daily load decision making. Journal of Environmental Engineering, 130, 594-604.

Karvonen, A., Rintamäki, P., Jokela, J., Valtonen, E.T. (2010). Increasing water temperature and disease risks in aquatic systems: Climate change increases the risk of some, but not all, diseases. International Journal for Parasitology, 40, 1483-1488.

Kattelus, M., Kummu, M., Keskinen, M., Salmivaara, A., Varis, O. (2015). China's southbound transboundary river basins: a case of asymmetry. Water International, 40(1), 113-138.

Kaufmann, P.R., Larsen, D.P., Faustini, J.M. (2009). Bed Stability and Sedimentation Associated With Human Disturbances in Pacific Northwest Streams. J. American Water Resources Association, 45(2), 434-459.

Kaufmann P.R., Hughes R.M., Van Sickle J., Whittier T.R., Seeliger C.W., Paulsen S.G. (2014). Lake shore and littoral habitat structure: a field survey method and its precision. Lake and Reservoir Management, 30(2), 157-176.

Keessen, A. M., Van Kempen, J. J. H. , Van Rijswick, H. F. M. W., Robbe, J., Backes, C. W. (2010). European river basin districts: are they swimming in the same implementation pool? Journal of Environmental Law, 22(2), 197-222.

Kelly, M., Juggins, S., Guthrie, R., Pritchard, S., Jamieson, J., Rippey, B., Hirst, H., Yallop, M. (2008). Assessment of ecological status in UK rivers using diatoms. Freshwater Biology, 53, 403-422.

Kemker, C. (2013a). "pH of Water." Fundamentals of Environmental Measures. Fondriest Environmental, Inc, 19 Nov. 2013. http://www.fondriest.com/environmental-measurements/parameters/water-quality/ph [Accessed 17 February 2016]

Kemker, C. (2013b). "Dissolved Oxygen." Fundamentals of Environmental Measurement. Fondriest Environmental, Inc, 19 Nov. 2013. http://www.fondriest.com/environmental-measurements/parameters/water-quality/dissolved-oxygen/ [Accessed 17 February 2016]

Kennard, M.J., Pusey, B.J., Olden, J.D., MacKay, S.J., Stein, J.L., Marsh, N. (2010). Classification of natural flow regimes in Australia to support environmental flow management. Freshwater Biology, 55(1), 171-193.

Kennedy, V.S. (2004). Thermal pollution. Encyclopedia of Energy, Volume 6, Elsevier academic Press

Kent, M., Coker, P. (1992). Vegetation description and analysis. John Wiley & Sons, London.

Kilgour, B. W., Keith M. S., David E. M. (1998). Using the normal range as a criterion for ecological significance in environmental monitoring and assessment. Ecoscience, 542-550.

King, L., Barkey, P., Jones, R.I. (2000). Epilithic algal communities and their relationship to environmental variables in lakes of the English Lake District. Freshwater Biology, 45, 425-442.

King, J.M., Tharme, R.E., De Villiers, M.S. (eds.) (2000). Environmental flow assessments for rivers: manual for the Building Block Methodology. Water Research Commission Technology Transfer Report No. TT131/00. Water Research Commission, Pretoria. 340 pp.

King, J. M., Tharme, R.E., de Villiers, M.S. (2008). Environmental flow assessments for rivers: Manual for the building block methodology. Report to the Water Research Commission, WRC Report. TT 354/08. Gezina, South Africa.

King, J.M., Brown, C.A. (2010). Integrated flow assessments: Concepts and method development in Africa and South East Asia. Freshwater Biology, 55, 127-146.

King, J., Beuster, H., Brown, C., Joubert, A. (2014). Pro-active management: the role of environmental flows in transboundary cooperative planning for the Okavango River system. Hydrological Sciences Journal, 59 (3–4), 1–15.

Kirillin, G., Shatwell, T., Kasprzak, P. (2013). Consequences of thermal pollution from a nuclear plant on lake temperature and mixing regime. Journal of Hydrology, 496, 47–56.

Klijn, E.-H., Steijn, B., Edelenbos, J. (2010). The Impact of network management on outcomes in governance networks. Public Administration, 88(4), 1063–1082.

Kolkwitz, R., Marsson, M. (1909). Ökologie der tierischen Saprobien. Beiträge zur Lehre von der biologischen Gewäasserbeurteilung. Internationale Revue der Gesamten Hydrobiologie und Hydrogeogrrapie, 2, 126-152. [Translated 1967]. Ecology of animal saprobia. In: Keup, L.E., Ingram, W.M., Machenthum, K.M. Biology of Water Pollution. Federal Water Pollution Control Administration, 485-495.

Komatsu, E., Fukushima, T., Harasawa, H. (2007). A modeling approach to forecast the effect of long-term climate change on lake water quality. Ecological Modelling, 209(2), 351-366.

Kottelat, M., Freyhof, J. (2007). Handbook of European freshwater fishes. Kottelat, Cornol and Freyhof, Berlin.

Kovacs, P., Buzas, Z., Laszlo, F., Kelnarova, Z., Durkovicova, D., Zimnikovalova, O. (1998). Report no. 1 (Inception Report): Pilot Project Programme Transboundary Rivers. Task Force on Monitorimng and Assessmente under the UNECE Water Convention. 40pp.

Kovacs, A., Clement, A. (2009). Impacts of the climate change on runoff and diffuse phosphorus load to Lake Balaton (Hungary). Water Science & Technology, 59(3), 417-423.

KPMG (2011). Department of Sustainability, Environment, Water, Population and Communities. Evaluation of the National water Quality Management Strategy.

Krause, S., Lewandowski, J., Dahm, C. N., Tockner, K. (2015). Frontiers in real-time ecohydrology–a paradigm shift in understanding complex environmental systems. Ecohydrology, 8(4), 529-537.

Kummu, M., de Moel, H., Ward, P. J., Varis, O. (2011). How close do we live to water? A global analysis of population distance to freshwater bodies. PLOS ONE, 6(6), e20578.

Kundzewicz, Z.W., Mata, L.J., Arnell, N.W., Döll, P., Jimenez, B., Miller, K., Oki, T., Sen, Z., Shiklomanov, I. (2008). The implications of projected climate change for freshwater resources and their management. Hydrological Sciences Journal, 53 (1), 3-10.

Kura, Y., Revenga, C., Hocino, E., Mock G. (2004). Fishing for Answers: Making sense of the global fish crisis. World Resources Institute, Washington, DC.

Kutas, T., Herodek, S. (1986). A complex model for simulating the Lake Balaton ecosystem. In: Modeling and Managing Shallow Lake Eutrophication. Springer Verlag Berlin and Heidelberg, 309-322.

Lake, P. S., Palmer, M. A., Biro, P., Cole, J., Covich, A. P., Dahm, C., Gibert, J., Goedkoop, W., Martens, K., Verhoeven, J. (2000). Global change and the biodiversity of freshwater ecosystems: Impacts on linkages between above-sediment and sediment biota. BioScience, 50, 1099–1107.

Láng, I. (1986). Impact on Policy Making: Background to a Government Decision. In: Somlyódy, L., van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Lang, C., Reymond, O. (1993). Recovery of Lake Neuchâtel (Switzerland) from eutrophication indicated by the oligochaete communities. Archiv für Hydrobiologie, 128, 65-71.

Lang, C., Reymond, O. (1996). Empirical relationships between oligochaetes, phosphorus and organic deposition during the recovery of Lake Geneva from eutrophication. Archiv für Hydrobiologie, s136, 237-245.

Laplace, P.S. (1985). Philosophical Essay on Probabilities. Springer New York. 292pp.

Lauritsen D.D., Mozley S.C., White D.S. (1985). Distribution of oligochaetes in Lake Michigan and comments on their use as indices of pollution. Journal of Great Lakes Research, 11, 67-76.

Lecointe, C., Coste, M., Prygiel, J. (1993). "Omnidia": software for taxonomy, calculation of diatom indices and inventories management. Hydrobiologia, 269/270, 509–513.

Leentvaar, J., (2015). Reader Water and Environmental Policy Making, UNESCO-IHE, Delft.

Lehner, B., R-Liermann, C., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P. et al.: High resolution mapping of the world's reservoirs and dams for sustainable river flow management. Frontiers in Ecology and the Environment. Source: GWSP Digital Water Atlas (2008). Map 81: GRanD Database (V1.0). Available online at http://atlas.gwsp.org.

Lehner, B., Liermann, C. R., Revenga, C., Vörösmarty, C., Fekete, B., Crouzet, P., Döll, P., Endejan, M., Frenken, K., Magome, J., Nilsson, C., Robertson, J.C., Rödel, R., Sindorf, N., Wisser, D. (2011). High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Frontiers in Ecology and the Environment, 9(9), 494-502.

Lehner, B. (2012). Hydrobasins: version 1. b. Global watershed boundaries and sub-basin delineation derived from hydrosheds data at 15 second resolution. World Wildlife Fund, Washington, DC.

Leira, M., Jordan, P., Taylor, D., Dalton, C., Bennion, H., Rose, N., Irvine, K. (2006). Assessing the ecological status of candidate reference lakes in Ireland using palaeolimnology. Journal of Applied Ecology, 43, 816-827.

Le Quesne T., Kendy E., Weston D. (2010). The implementation challenge. Taking stock of governmental policies to protect and restore environmental flows. The Nature Conservancy and WWF report. 67 pp.

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T. M., Bertzky, B., Butchart, S.H., Stuart, S.N., Badman, T., Rodrigues, A. S. (2013). Conservation. Protected areas and effective biodiversity conservation. Science, 342(6160), 803-805.

Lewis, S.L. (2012): We must set planetary boundaries wisely. Nature 485, page 417. http://www.nature.com/news/we-must-set-planetary-boundaries-wisely-1.10694[Accessed 18 February 2016]

Liess, M., Schäfer, R. B., Schriever, C. A. (2008). The foot print of pesticide stress in communities—Species traits reveal community effects of toxicants. Science of the Total Environment, 406, 484-490.

Lijklema, L., Gelencsér, P., Szilágyi, F., Somlyódy, L. (1986): Sediment and its Interaction with Water. In: Somlyódy, L., and van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Lindenmayer, D.B., Likens, G.E. (2009). Adaptive monitoring: A new paradigm for long-term research and monitoring. Trends in Ecology & Evolution, 24(9), 482-486.

Lindenmayer, D.B., Likens, G. E. (2010a). Effective ecological monitoring. CSIRO Publishing, Melbourne and Earthscan, London.

Lindenmayer, D., Likens, G. (2010b), 'The science and application of ecological monitoring', Biological Conservation, 143(6), 1317-132.

Linke, S., Norris, R., Faith, D. (2002). Australian River Assessment System: Improving AusRivAS Analytical Methods DDRAM and E-Ball (Phase I Final Report), Monitoring River Heath Initiative Technical Report no 27, Australian Museum, Sydney, Commonwealth of Australia, Canberra and University of Canberra, Canberra.

Linke, S., Tutak, E., Nel, J. (2011). Freshwater conservation planning: the case for systematic approaches, Freshwater Biology, 56, 6-20.

Loh, J., Green, R.H., Ricketts, T., Lamoreux, J., Jenkins, M., Kapos V., Randers, J. (2005). The Living Planet Index: Using species population time series to track trends in biodiversity. Philosophical Transactions of the Royal Society B, 260, 289-295.

Lovett, G.M., Burns, D.A., Driscoll, C.T., Jenkins, J.C., Mitchell, M.J., Rustad, L., Shanley, J.B., Likens, G.E., Haeuber, R. (2007). Who needs environmental monitoring?. Frontiers in Ecology and the Environment, 5(5), 253-260.

Luettich, R. A. and Harleman, D.R.F. (1986). A Comparison of Water Quality Models and Load Reduction Predictions. In: Somlyódy, L., and van Straten, G., (eds.): Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Lugwisha, R.H.J., Leentvaar, J., Baya, B.T., Douven, W.J.A.M. (2008). Challenges on compliance and enforcement of the wastewater management legislation in Tanzania, Proceedings of the 8th International INECE conference, Cape Town, South Africa.

Luoma, S.N. (1983). Bioavailability of trace metals to aquatic organisms—A review. The Science of the Total Environment, 28, 1-22.

Lyons, J., Navarro-Perz, S., Cochran, P.A., Samtana, C.E., Guzman-Arroyo, M. (1995). Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in west central Mexico. Conservation Biology, 9, 569-584.

Macfarlane, D., Kotze, D., Ellery, W., Koopman, V.. Goodman, P.S., Goge M., Walters D. (2009). WET-Health. http://www.wrc.org.za/Pages/DisplayItem.aspx?ItemID=9025&FromURL=%2FPages%2FDefault.aspx%3F [Accessed 17 February 2016]

Macuiane, A. M., Hecky, R.E., Guildford, S.J. (2015). Changes in fish community structure associated with cage aquaculture in Lake Malawi, Africa. Aquaculture, 448: 8–17.

Maguire, C., Gibson, C. (2005). Ecological change in Lough Erne: influence of catchment changes and species invasions. Freshwater Forum, 24 (1).

Malmqvist, B., Rundle, S. (2002). Threats to the running water ecosystems of the world. Environmental Conservation, 29(02), 134-153.

Maltby, E. (2009). Functional assessment of wetlands : towards evaluation of ecosystem services. Woodhead Publishing, Cambridge.

Martuzzi, M., Tickner, J.A. (2004). The precautionary principle: protecting public health, the environment and the future of our children. World Health Organization. http://www.euro.who.int/__data/assets/pdf_file/0003/91173/E83079.pdf [Accessed 02 February 2016]

Matthews, J. H., Le Quesne, T., Wilby, R., Pegram, G., Von der Heyden, C., Wickel, A. J., Hartmann, J., McSweeney, C., Guthrie, C., Blate, G., Kimura de Frietas, G., Levine, E. (2009). Flowing forward: Informing climate-resilient biodiversity conservation and integrated water resources management decisions. The World Bank, Washington, DC.

Matthews, J.H., Wickel, A.J. (2009). Embracing uncertainty in freshwater climate change adaptation: A natural history approach. Climate and Development, 1, 269-279.

Matthews, J. H., Forslund, A., McClain, M.E., Tharme R.E. (2014). More than the fish: environmental flows for good policy and governance, poverty alleviation and climate adaptation. Aquatic Procedia 2, 16-23.

McDowell, R., Trudgill, S. (2000). Variation of phosphorus loss from a small catchment in south Devon, UK. Agriculture, Ecosystems and Environment, 79, 143-157.

MDBA (Murray-Darling Basin Authority) (2012). Basin Plan. https://www.comlaw.gov.au/Details/F2012L02240 [Accessed 02 February 2016]

MDEP (Maine Department of Environmental Protection) (2012a). 2012 Integrated Water Quality - Monitoring and Assessment Report; DEPLW-1246; 219 pages http://www.maine.gov /dep/water/monitoring/305b/ [Accessed 02 February 2016]

MDEP (Main Department of Environmental Protection) (2012b). Maine Impervious cover total Maximum Daily Load Assessment for impaired streams (TMDL). August, Maine, USA http://www.maine.gov/dep/water/monitoring/tmdl/2012/IC%20TMDL_Sept_2012.pdf [Accessed 02 February 2016]

MDEP (Main Department of Environmental Protection) (2012c). Total Maximum Daily Load Assessment (TMDL) for Impaired Streams. DEPLW-1239. Maine Department of Environmental Protection, Augusta, Maine. http://www.maine.gov/dep/water/monitoring/tmdl/2012/IC%20TMDL_Sept_2012.pdf [Accessed 02 February 2016]

MEA (Millennium Ecosystem Assessment) (2003). Ecosystems and Human Wellbeing: a Framework for Assessment. Island Press, Washington, DC.

MEA (Millennium Ecosystem Assessment) (2005a). Introduction and conceptual framework. Ecosystems and human well-being: a framework for assessment. Island Press, Washington, D.C.

MEA (Millennium Ecosystem Assessment) (2005b). Ecosystems and Human Well-being: Synthesis. Island Press, Washington, DC.

Meybeck M., Chapman D.V., Helmer R. (1989). Global Freshwater Quality: A first assessment. Blackwell Reference, Oxford.

Meybeck, M., Mikstatch, V., Helmer. R. (1996). Strategies for Water Quality Assessment. In: Champan D.V. (ed.). Water Quality Assessments: A guide to the use of biota, sediments and water in environmental monitoring. 2nd edition, Chapman & Hall, London.

Meyer, J.L. (1997). Stream Health: Incorporating the human dimension to advance stream ecology. Journal of the North American Benthological Society, 16, 439–447.

Meyer, L., Du Preez, L., Bonneau, E., Héritier, L., Quintana, M.F., Valdéon, A., Sadaoui, A., Kechemir-Issad, N., Palacios, C., Verneau, O. (2015). Parasite host-switching from the invasive American red-eared slider, Trachemys scripta elegans, to the native Mediterranean pond turtle, Mauremys leprosa, in natural environments. Aquatic Invasions, 10(1), 79-91.

Milbrink, G. (1978). Indicator communities of oligochaetes in Scandanavian lakes. Verhandlungen Internationale Vereinigung für theoretische und angewandte Limnologie, 20, 2406-2411.

Miler, O., Porst, G., McGoff, E., Pilotto, F., Donohue, L., Jurca, T., Solimini, A., Sandin, L., Irvine, K., Aroviita, J., Clarke, R., Pusch, M. T. (2013). Morphological alterations of lakeshores in Europe: A multimetric ecological assessment approach using benthic macroinvertebrates, Ecological Indicators, 34, 398-410.

Mills, L. J., Chichester, C. (2005). Review of evidence: Are endocrine-disrupting chemicals in the aquatic environment impacting fish populations?. Science of the Total Environment, 343(1), 1-34.

Milly, P.C.D., Betancourt, J., Falkenmark, M., Hirsch, R.M., Kundzewicz, Z.W., Lettenmaier, D.P., Stouffer R. J. (2008). Stationarity Is Dead: Whither Water Management? Science, 319, 573-574.

Mittermeier, R.A., Farrell, T.A., Harrison, I.J., Upgren, a.J., Brooks, T.M. (eds.) (2010). Freshwater- The essence of Life, Conservation International and CEMEX, Washington, DC.

MoE (Ministry of the Environment, Indonesia) (2001). Indonesian Government Regulation No.82 of 2001. Regulation of water quality management and water pollution control.

MoE (Ministry of the Environment, Indonesia) (2003). Decree of Ministry of the Environment, Indonesia No. 115 of 2003. Evaluation of ecosystem status criteria based on national water quality standard.

MoE (Ministry of the Environment, Indonesia) (2008). Guideline for lake management.

MoE (Ministry of Environment) (2010). National lake priority 2010 – 2014. Ministry of Environment, 2010

MoE (Ministry of Environment Indonesia) (2009). Environmental Protection and Management. Law No 32 of 2009 http://faolex.fao.org/docs/pdf/ins97643.pdf [Accessed 16 February 2016]

MoE (Ministry of the Environment, Indonesia) (2011). Report of evaluation of government regulation No. 82 of 2001 on water quality management and water pollution control.

MoE (Ministry of the Environment, Indonesia) (2012). State of the Environment Report Indonesia. Pillars of the environment Indonesia. Republic of Indonesia.

MoE (Ministry of Environment, Indonesia) (2015). A Draft of Ministrial Decree Concerning Management of Inland Water Ecosystem Impairment. Republic of Indonesia.

MoEJ (1997). Environmental quality standards for water pollution. Ministry of the Environment, Japan. http://www.env.go.jp/en/water/wq/wp.pdf [Accessed 16 February 2016]

MoEJ (Ministry of the Environment of Japan) (2012). WEPA Outlook on Water Environmental Management in Asia. Ministry of the Environment, Japan, 2012.

Mohseni, O., Stefan, H. G., Eaton, J. G. (2003). Global warming and potential changes in fish habitat in US streams. Climatic Change, 59(3), 389-409.

Molur, S., Smith, K.G., Daniel, B.A., Darwall, W.R.T. (compilers) (2011). The Status and Distribution of Freshwater Biodiversity in the Western Ghats, India. International Union for Conservation of Nature, Gland and Zoo Outreach Organisation, Coimbatore.

Moog, O., Schmidt-Kloiber, A., Ofenböck, T. , Gerritsen, J. (2004). Does the ecoregion approach support the typological demands of the EU Water Framework Directive?. Hydrobiologia, 516, 21-33.

Morse, J.C., Bae, Y.J., Munkhjargal, G., Sangpradub, N., Tanida, K., Vshivkova, T. S., Wang, B., Yang, L., Yule, C.M. (2007). Freshwater biomonitoring with macroinvertebrates in East Asia. Frontiers in Ecology and the Environment, 5(1), 33-42.

Moschini-Carlos, V, Bortoli, S, Pinto, E. Nishimura, P.Y, Freitas, L.G Pompêo, M.L.M, Dorr, F. (2009). Cyanobacteria and Cyanotoxin in the Billings Reservoir (São Paulo, SP, Brazil). Limnetica, 28 (2), 273-282.

Moss, B. (1983).The Norfolk Broadland: Experiments in the restoration of a complex wetland. Biological Reviews, 58, 521-561.

Moss, B., Stephen, D., Alvarez, C., Becares, E., van den Bund, W., Collings, S.E., van Donk, E., de Eyto, E., Feldmann, T., Fernández-Aláez, C., Fernández-Aláez, M., Franken, R.J.M, García-Criado, F., Gross, E.M., Gyllström, M., Hansson, L-A., Irvine, K., Järvalt, A., Jensen, J-P., Jeppsen, E., Kairesalo, T., Kornijów, R., Krause, T., Künnap, H., Laas, A., Lill, E., Lorens, B., Luup, H., Miracle, M.R., Nõges, P., Nõges, T., Ott, I., Peczula, W., Peeters, E.T.H.M., Phillips, G., Romo, S., Russell, V., Salujõe, J., Scheffer, M., Siewertschen, K., Smal, H., Tesch, C., Timm, H., Tuvikene, L., Tonno, I., Virro, T., Vicente, E., Wilson, D. (2003). The determination of ecological quality in shallow lakes - a tested expert system (ECOFRAME) for implementation of the European Water Framework Directive. Aquatic Conservation: Marine and Freshwater Systems, 13, 507-550.

Moss, B. (2008). The Water Framework Directive: Total environment or political compromise? Science of the Total Environment, 400, 32-41.

Moss, T. (2012). Spatial fit, from panacea to practice: implementing the EU Water Framework Directive. Ecology and Society, 17(3), 2.

Murray-Bligh, J., Ferguson, A. (1999). Procedures for collecting and analysing macroinvertebrate samples. Environment Agency Report, 2nd Edition BT001.

Naiman, R.J., Bilby, R.E. (1998). River Ecology and Management: Lessons from the Pacific Coastal Eco region. Springer Verlag, New York.

Nakamura, K., Tockner, K., Amano, K. (2006). River and wetland restoration: Lessons from Japan. BioScience, 56(5), 419-429.

Naumann, E. (1931). Limnologische Terminologie. Urban und Schwarzenberg, Berlin. Nel, J.L., Roux, D.J., Abell, R., Ashton, P.J., Cowling, R.M., Higgins, J.V., Thieme, M., Viers J. H. (2009). Progress and challenges in freshwater conservation planning. Aquatic Conservation: Marine and Freshwater Ecosystems, 19, 474-485.

Neill, C., Deegan, L.A., Thomas, S.M., Haupert, C.L., Krusche, A.V., Ballester, M.V.R., Victoria, R.L. (2006). Deforestation alters the hydraulic and biogeochemical characteristics of small lowland Amazonian streams. Hydrological Processes, 20, 2563–2580.

Nel, J.L., Roux, D.J., Abell, R., Ashton, P.J., Cowling, R.M., Higgins, J.V., Thieme, M., Viers, J. H. (2009), Progress and challenges in freshwater conservation planning. Aquatic Conservation: Marine and Freshwater Ecosyst.ems, 19, 474–485.

Newig, J., Günther, D., and Pahl-Wostl, C. (2010). Synapses in the network: Learning in governance networks in the context of environmental management. Ecology and Society, 15(4).

Newman, G., Wiggins, A., Crall, A., Graham, E., Newman, S., Crowston, K. (2012). The future of citizen science: emerging technologies and shifting paradigms. Frontiers in Ecology and the Environment, 10(6), 298-304.

Newson, M.D., Large, A.R.G. (2006). 'Natural' rivers, 'hydromorphological quality' and river restoration: a challenging new agenda for applied fluvial geomorphology. Earth Surface Process and Landforms, 31, 1606–1624.

Nietch, C.T., Lazorchak, J. M., Johnson, B., Allen, J. (2014). Evaluation of total dissolved solids and specific conductance water quality targets with paired single-species and mesocosm community exposures. Society of Environmental Toxicology and Chemistry. Vancouver, BC, Canada.

Nijboer, R.C., Verdonschot, P.F.M. (2004). Rate and common macroinvertebrates: definition of distribution classes and their boundaries. Archives fur Hydrobiologia, 161, 45-64.

Nijboer, R.C., Verdonschot, P.F.M., van der Werf, D.C. (2005). The use of indicator taxa as representatives of communities in bioasesssment. Freshwater Biology, 50, 1427-1440.

Nilsson, J., Grennfelt, P. (1988). Critical loads for Sulphur and Nitrogen. Nordic Council of Ministers, Copenhagen, pp. 418.

Nilsson, C., Reidy, C.A., Dynesius, M., Revenga, C. (2005). Fragmentation and Flow Regulation of the World's Large River Systems. Science, 308, 405-408.

Nilsson, C. Renöfält, B.M. (2008). Linking flow regime and water quality for rivers: a challenge to adaptive catchment management. Ecology and Society 13, 18.

Noble, R., Cowx, I. (2002). Final Report. Development, Evaluation & Implementation of a Standardised Fish-based Assessment Method for the Ecological Status of European Rivers - A Contribution to the Water Framework Directive (FAME). Development of a river-type classification system (D1); Compilation and harmonization of fish species classification (D2). University of Hull, UK.

Nõges, P., van de Bund, W., Cardoso, A.C., Solimini, A.G., and Heiskanen, A.S. (2009). Preface: Assessment of the Ecological Status of European Surface Waters. Hydrobiologia, 633, 1-3.

Noon, B.R., Spies, T.A., Raphael. M.G. (1998). Conceptual Basis for Designing an Effectiveness Monitoring Program, Chapter 2. In: The Strategy and Design of the Effectiveness Monitoring Programme for the Northwest Forest Plan, General Technical Report PNW-GTR-437, Portland, Or: USDA Forest Service Pacific Northwest Research Station, 21-48.

Noon, B.R., Spies, T.A., Raphael, M.G. (1999). Conceptual basis for designing an effectiveness monitoring program. In: Mulder, B.S., et al. (Eds.), The Strategy and Design of the Effectiveness Monitoring Program for the Northwest Forest Plan. US Department of Agriculture, Forest Service, Gen. Tech. Rep. PNW-GTR-437, Portland, OR, pp. 21–48.

Nordhaus, T., Shellenberger, M., Blomqvist, L. (2012). The planetary boundaries hypothesis: a review of the evidence. Breakthrough Institute, Oakland USA, 42 pages. http://thebreakthrough.org/blog/Planetary%20Boundaries%20web.pdf. [Accessed 18 February 2016]

Norris, R.H., Hawkins, C.P. (2000). Monitoring river health. Hydrobiologia, 435, 5–17.

Norton, S.B., Cormier, S.M., Suter, G.W. (2015). Ecological Causal Assessment. CRC Press, Taylor and Francis, Boca Raton, FL, USA.

Novotny, V., Somlyody, L. (eds.) (1995). Remediation and Management of Degraded River Basins with Emphasis on Central and Eastern Europe. Springer Verlag, Heidelberg.

NRCD (National River Conservation Directorate) (2008). Guidelines for National Lake Conservation. Ministry of Environment & Forest. National River Conservation Directorate, Paryavaran Bhawan, New Dehli.

NSW Government (New South Wales Government) (2010). Assessing the extent and condition of wetlands in NSW. Monitoring, evaluation and reporting program. Technical report series. State of NSW and Office of Environment and Heritage, Sydney.

NWQMC (National Water Quality Monitoring Council) (2006). A national water quality monitoring network for US coastal waters and their tributaries. Final Report to the President's Council on Environmental Quality, the National Science and Technology Council's Subcommittee on Water Availability and Quality, and the Joint Subcommittee on Ocean Science and Technology. http://acwi.gov/monitoring/network/design.

NWQMS (National Water Quality Management Strategy) (1998). Implementation Guidelines. http://www.environment.gov.au/system/files/resources/60a23a28-ebb4-4562-bf31-8166d45b1fda/files/nwqms-implementation-guidelines.pdf

O'Hare, M.T., Tree, A., Neale, M.W., Irvine, K., Gunn, I.D., Jones, J.I., Clarke, R.T. (2004). Review of Lake Benthic Macro-invertebrate Sampling Methods and Strategy. R & D Technical Report 13765. Phase 1 report to the U.K. Environment Agency.

OECD (Organisation for Economic Co-operation and Development) (1982). Eutrophication of waters. Monitoring, assessment and control. OECD (Organisation for Economic Co-operation and Development) (2011), Water Governance in OECD Countries: A Multi-level Approach, OECD Studies on Water, OECD Publishing, Paris, France.

OECD (Organisation for Economic Co-operation and Development) (1992). Report of the OECD workshop on extrapolation of laboratory aquatic toxicity data to the real environment. OECD Environment monographs No.59. OECD, Paris, France.

OECD (Organisation for Economic Co-operation and Development) (1993). OECD core set of indicators for environmental performance reviews. OECD Environment Monographs No. 83. OECD, Paris, France.

OECD (Organisation for Economic Co-operation and Development) (1995). Guidance document for aquatic effects assessment. OECD Environment monographs No.92. OECD, Paris, France.

OECD (Organisation for Economic Co-operation and Development) (2011). Water Governance in OECD Countries: A Multi-level Approach, OECD Studies on Water, OECD Publishing, Paris, France.

O'Keeffe, J., Le Quesne, T. (2009). Keeping Rivers Alive - A primer on environmental flows, WWF Water Security Series 2.

Olden, J.D., Kennard, M.J., Pusey, B.J. (2008). Species invasions and the changing biogeography of Australian freshwater fishes. Global Ecology and Biogeography, 17, 25-37.

Ollis, D. J., Dallas, H.F., Esler, K.J., Bocher, C. (2006). Rapid bioassessment of the ecological integrity of river ecosystems using aquatic macroinvertebrates: review with a focus on South Africa, African Journal of Aquatic Science, 31(2), 205-227.

O'Neill, B.J., Thorp, J.H. (2011). A simple channel complexity metric for analyzing river ecosystem responses. River Systems, 19(4), 327-335.

Opperman, J., Grill, G., Hartmann, J. (2015). The Power of Rivers: Finding balance between energy and conservation in hydropower development. The Nature Conservancy: Washington, D.C.

O'Riordan, T. (2004). Environmental science, sustainability and politics in Transactions of the Institute of British Geographers, 29, 234-247.

Ormerod, S.J., Dobson, M., Hildrew, A.G., Townsend, C.R. (2010). Multiple Stressors in Freshwater Ecosystems. Freshwater Biology, 5(1), 1-269.

Ollis, D. J., Dallas, H.F., Rieradevall, M., Prat, N., Day, J.A. (2006) Rapid bioassessment of the ecological integrity of river ecosystems using aquatic macroinvertebrates: review with a focus on South Africa, African Journal of Aquatic Science, 31(2), 205-227.

Olson, J. R., Hawkins, C.P. (2012). Predicting natural base-flow stream water chemistry in the western United States, Water Resour. Res., 48, W02504.

O'Toole, C., Irvine, K. (2006). Responses of aquatic biota to combined pressures from nutrients and priority substances. In: Solimini, A.G., Cardosa, A. C., Heiskanen, A-S. (eds.). Indicators and methods for the ecological status assessment under the Water Framework Directive. European Commission, Joint Research Centre, Institute for Environment and Sustainability, 225-248.

Padisák, J. (2002). Use of phytoplankton to characterize the biological state of lakes according to the EU Water Famework Directive. Background Report, Deptartment of Sanitary and Environmental Engineering, Budapest University of Technology. (In Hungarian).

Pagenkopf, G.K. (1976). Zinc speciation and toxicity to fishes. Proceedings of Toxicity to Biota of Metal Forms in Natural Water (ISC).

Pagenkopf, G.K. (1983). Gill surface interaction model for trace-metal toxicity to fishes: role of complexation, pH, and water hardness. Environmental Science and Technology, 17(6), 342-347.

Pahl-Wostl, C. (2002). Towards sustainability in the water sector – The importance of human actors and processes of social learning. Aquatic Sciences, 64, 394–411.

Pahl-Wostl, C. (2007). The implications of complexity for integrated resources management. Environmental Modelling and Software, 22, 561–569.

Pahl-Wostl, C., J. Sendzimir, P. Jeffrey, J. Aerts, G. Berkamp, and K. Cross. (2007). Managing change toward adaptive water management through social learning. Ecology and Society, 12(2), 30.

Pahl-Wostl, C., Mostert, E., Tabara, D. (2008). The Growing Importance of Social Learning in Water Resources Management and Sustainability Science. Ecology and Society, 13(1), p.24.

Pahl, Wostl, C., Lebel, L., Knieper, C., Nikita, E. (2012). From applying panaceas to mastering complexity: towards adaptive water governance in river basins. Environmental Science and Policy, 23, 24-34.

Pahl-Wostl, C., Arthington, A., Bogardi, J., Bunn, S., Hoff, H., Lebel, L., Nikitina, E., Palmer, M., Poff, L. N., Richards, K., Schluter, M., Schulze, R., St-Hilaire, A., Tharme, R., Tockner, K., Tsegai, D. (2013). Environmental flows and water governance: managing sustainable water uses. Current Opinion in Environmental Sustainability, 5, 341-351.

Palaniappan, M., Gleick, P.H., Allen, L., Cohen, M.J., Christian-Smith, J., Smith, C., Ross, N. (2010).Clearing the Waters: A focus on water quality solutions. United Nations Environment Program, 91p.

Palmer, M.A., Menninger, H.L., Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice?. Freshwater Biology, 55(s1), 205-222.

Palmer, C. G. (1999). The application of ecological research in the development of a new water law in South Africa. , Journal of the North American Benthological Society, 18, 132-142.

Palmer, C.G., Muller, W.J., Gordon, A.K., Scherman, P.A., Davies-Coleman, H.D., Pakhomova, L., de Kock E. (2004a). The development of a toxicity database using freshwater madcroinvertebrates, and its application to the protection of South African water resources. South African Journal of Science 100(11-12), 643-650.

Palmer, C.G., Muller, W.J., Davies-Coleman, H. (2004b). Applied environmental water quality. Proceedings of the 2004 Water Institute of Southern Africa (WISA) Biennial Conference, May 2-6 2004, Cape Town, South Afirca, pp. 273-279.

Palmer, C. G., Rossouw, N., Muller, W.J., Scherman, P.-A. (2005). The development of water quality methods within ecological Reserve assessments, and links to environmental flows. Water SA, 31(2), 161-170.

Palmer, M.A., Menninger, H.L., Bernhardt, E. (2010). River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice?. Freshwater Biology, 55(s1), 205-222.

Papoutsa, C., Hadjimitsis, D.G. (2013). Remote Sensing for Water Quality Surveillance in Inland Waters: The Case Study of Asprokremmos Dam in Cyprus, Remote Sensing of Environment - Integrated Approaches, ISBN: 978-953-51-1152-8, InTech, DOI: 10.5772/39308. http://www.intechopen.com/books/remote-sensing-of-environment-integrated-approaches/remote-sensing-for-water-quality-surveillance-in-inland-waters-the-case-study-of-asprokremmos-dam-in. [Accessed 02 February 2016]

Pardo, I., Poikane, S., P., Bonne, W., Uszko, W., van de Bund, W., Owen, R., Kelly, M., Pont, D., Birk, S., Bennett, C., Gómez-Rodríguez, C., Wolfram, W., Ecke, F., van den Berg, M., Ritterbusch, R., Phillips, G., Brucet, S., Ortiz-Casas, J. (2010). Revision of the consistency in Reference Criteria application in the phase one of the European Intercalibration Exercise Version: Final version (18.01.10), GIG working Group on Reference Conditions. European Commission, Brussels.

Parsons, M., Norris, R. (1996). The effect of habitat-specific sampling on biological assessment of water quality using a predictive model. Freshwater Biology, 36, 419-434.

Parsons, M., Thoms, M., Flotemersch, J.E., Reid M. (2016). Monitoring the resilience of rivers as social-ecological systems: a paradigm shift for river assessment in the twenty-first century. In: Gilvear, D., Greenwood, M., Thoms, M., Wood, P. (eds.). River Systems: Research and Management for the 21st Century. Wiley & Sons, Ltd., New Jersey.

Partzsch, L. & Ziegler, R. (2011). Social entrepreneurs as change agents: a case study on power and authority in the water sector. International Environmental Agreements: Politics, Law and Economics, 11(1), pp.63–83.

PEAK (Porgera Environmental Advisory Komiti) (2009). Strickland River Report Card 2009. http://www.barrick.com/files/porgera/PEAK-Porgera-Report-Card-2010.pdf [Accessed 16 February 2016]

Pegram, G., Li, Y., Le. Quesne, T., Speed, R., Li, J., F. Shen, F. (2013). River basin planning: Principles, procedures and approaches for strategic basin planning. Paris, UNESCO. http://unesdoc.unesco.org/images/0022/002208/220866e.pdf [Accessed 02 February 2016]

Peh, K.S.H., Balmford, A., Bradbury, R.B., Brown, C., Butchart, S.H., Hughes, F.M., Stattersfield, A., Thomas, D.H.L., Walpole, M., Bayliss, J., Gowing, D., Jones, J.P.G., Lewis, S.L., Mulligan, M., Pandeya, B., Stratford, C., Thompson, J.R., Turner, K., Vira, B., Willcock, S., Birch, J.C. (2013). TESSA: a toolkit for rapid assessment of ecosystem services at sites of biodiversity conservation importance. Ecosystem Services, 5, 51–57.

Peraturan Bupati Agam Nomor 22 Tahun 2009. Pengelolaan Danau Maninjau. Lubuk Basung, Februari, 2009, 10 p.

Peterson, E.E., Sheldon, F., Darnell, R., Bunn, S.E. and Harch, B.D. (2011). A comparison of spatially explicit landscape representation methods and their relationship to seasonal stream conditions. Freshwater Biology 56, 590-610.

Pfeiffer, E., Leentvaar, J. (2013). Knowledge leads, policy follows? Two Speeds of Collaboration in River Basin Management. Water Policy, 15, 282-299.

Pikitch, E.K., Doukakis, P., Lauck L., Chakrabarty P., Erickson, D.L. (2005). Status, trends and management of sturgeon and paddlefish fisheries. Fish and Fisheries, 6, 233-265.

Pilgrim, E.M., Jackson, S.A., Swenson, S., Turcsanyi, I., Friedman, E., Weigt, L., Bagley, M.J. (2011). Incorporation of DNA barcoding into a large-scale biomonitoring program: opportunities and pitfalls. Journal of the North American Benthological Society, 30(1), 217-231.

Pimentel, D., Zuniga, R., Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics, 52, 273-288.

Pinel-Alloul, B., Méthot, G., Lapierre, L. and Willsie, A. (1996). Macroinvertebrate community as a biological indicator of ecological and toxicological factors in Lake Saint-François (Québec). Environmental Pollution, 91, 65-87.

PJV (Porgera Joint Venture) (2003). Porgera's Impact on Rivers. http://www.peakpng.org/resources/Porgeras_Impact_on_Rivers.pdf [Accessed 9 March 2016]

Playle RC, Dixon DG, Burnison K. (1993). Copper and cadmium binding to fish gills: Modification by dissolved organic carbon and synthetic ligands. Canadian Journal of Fisheries and Aquatic Sciences, 50, 2667–2677.

Playle, R.C., Dixon, D.G., Burnison, K. (1993). Copper and cadmium binding to fish gills: estimates of metal-gill stability constants and modelling of metal accumulation. Canadian Journal of Fisheries and Aquatic Sciences, 50, 2678–2687.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R.E., Stromberg, J. C. (1997). The natural flow regime. BioScience, 47(11), 769-784.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, P.B., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E., Warner, A. (2009). The ecological limits of hydrologic alteration (ELOHA): A new framework for developing regional environmental flow standards. Freshwater Biology. Freshwater Biology, 55(1), 147-170.

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobsen, R.B., Kennen, J.G., Merritt, D.M., O'Keeffee, J.H., Olden, J.D., Rogers, K., tharme, R.E., Warner, A. (2010). The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshwater Biology, 55(1), 147-170.

Poikane, S. (2009). Water Framework Directive intercalibration technical report. Part 2: Lakes. Joint Research Centre Institute for Environment and Sustainability, European Commission. http://www.apambiente.pt/dqa/assets/intercalibration-2003-2007-technical-report---lakes.pdf [Accessed 19 February 2016]

Pont, D., Hugueny, B., Beier, U., Goffaux, D., Melcher, A., Noble, R.C., Rogers, C., Roset, N., Schmutz, S. (2006). Assessing river biotic condition at the continental scale: a European approach using functional metrics and fish assemblages. Journal of Applied Ecology, 43, 70–80.

Pont, D., Hughes, R.M., Whittier, T.R., Schmutz, S. (2009). A Predictive Index of Biotic Integrity Model for Aquatic-Vertebrate Assemblages of Western US Streams. Transactions of the American Fisheries Society, 138, 292-305.

Porto, M. (2003). Recursos Hídricos e Saneamento na Região Metropolitana de São Paulo: Um desafio do tamanho da cidade. Agua Brasil series, no. 3. Brazil Water Resources. World Bank Group, Washington, DC.

Postel, S., Richter, B. (2003). Rivers for Life: Managing Water for People and Nature. Island Press, Washington, DC.

Posthuma L., Suter, G.W. II, Traas, T.P. (eds.) (2002). Species Sensitivity Distributions in Ecotoxicology. Lewis Publishers, Boca Raton, FL, USA.

Pringle, C.M. (1997). Exploring how disturbance is transmitted upstream: going against the flow. Journal of the North American Benthological Society, 16, 425–438.

Priscoli, J.D. (2006) River Basin Organizations http://www.transboundarywaters.orst.edu/research/case_studies/Documents/RiverBasinOrganizations.pdf [Accessed 02 February 2016]

Prygiel, J., Carpentier, P., Almeida, S., Coste, M., Druart, J.C. Ector, L., Guillard, D., Honoré, M.A. Iserentant, R., Ledeganck, P., Lalanne-Cassou, C., Lesinak. C., Mercier, I., Moncaut, P., Nazart, M., Nouchet, N., Peres, F., Peeters, V., Rimet, F., Rumeau, A., Sabater, S., Straub, F., Torrisi, M., Tudesque, L., Van Der Vijver, B., Vidal, H., Vizinet, J., Zydek, N. (2002). Determination of the Diatom Index (IBD NF T 90-354): results of an intercalibration exercise. Journal of Applied Phycology, 14, 27-39.

Puhspam, K. (2012). The Economics of Ecosystems and Biodiversity: Ecological and EconomicFoundations. Ramsar handbook. Routledge. http://www.ramsar.org. [Accessed 02 February 2016]

Raadgever, G.T. and Mostert, E., (2005). Transboundary River Basin Management – State-of-the-art review on transboundary regimes and information management in the context of adaptive management, Deliverable 1.3.1 of the NeWater project, RBA Centre, TU Delft.

Rahel, F.J. (2000) Homogenization of fish faunas across the United States. Science, 288, 854–856.

Rapport, D.J., Reigier, H.A., Hutchinson, T.C. (1985). Ecosystem Behaviour under Stress. American Naturalist, 125, 617-640.

Rapport, D.J., Costanza, R., McMichael, A.J. (1998). Assessing ecosystem health. Trends in Ecology and Evolution, 13, 397-402.

Raven, P.J., Holmes, N.T.H., Dawson, F.H., Everard, M. (1998). Quality assessment using river habitat survey data. Aquatic Conservation: Marine and Freshwater Ecosystems, 8, 477-499.

Raymond, C.M., Singh, G.G., Benessaiah, K., Bernhardt, J.R., Levine, J., Nelson, H., Turner, N.J., Norton, B., Tam, J., Chan, K.M. (2013). Ecosystem services and beyond: Using multiple metaphors to understand human–environment relationships. BioScience, 63(7), 536-546.

RCL (Research Center for Limnology, Indonesian Institute of Sciences) (2001). Limnological Characteristics of Lake Maninjau, Indonesia. Technical report.

RCL (Research Center for Limnology, Indonesian Institute of Sciences) (2002). Scientific recommendations (Academic Script) of Lake Maninjau.

RCL (Research Center for Limnology, Indonesian Institute of Sciences) (2014). Scientific recommendations (Academic Script) of Lake Maninjau.

RCL (Research Center for Limnology, Indonesian Institute of Sciences) (2016). Sistem Informasi Danau Indonesia (SIDI). http://danau.limnologi.lipi.go.id [Accessed 17 February 2016]

Reichert, P., Borsuk, M., Hostman, M., Schweizer, S., Spörri, C., Tockner, K., Truffel. B. (2007). Concepts of decision support for river rehabilitation, Environmental Modelling & Software, 22, 188-201.

Reitberger, B., Matthews, J., Feld, C., Davis, W., Palmer, M. (2010). Comparison of Monitoring Systems to Assess Ecological Status of Rivers in the US and Europe with Regard to the System's Applicability to Assess Success and Restoration / Rehabilitation. Final Report Jointly Submitted to the German EPA (Umweltbundesamt) and US EPA.

Renn, O. (2008). The Precautionary Approach to Risk Analysis. In: Risk Governance: coping with uncertainty in a complex world, Earthscan London, 79-92.

Ramsar Convention (2016). The Ramsar Convention Manual. 6th Ed.. http://www.ramsar.org/ [Accessed 19 February 2016]

Republic of Kenya (2002). The Water Act, 2002, No. 8 of 2002.

REFORM. (2016). Restoring rivers for effective catchment management. http://www.reformrivers.eu/.[Accessed 19 February 2016]

Resh, V.H., Jackson, J.K. (1993). Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In: Rosenberg, D.M., Resh, V.H. (eds.). Freshwater Biomonitoring and Benthic Macroinvertebrates. Chapman and Hall, London, 195 - 233.

Reynoldson, T. B., Norris, R. H., Resh, V. H., Day, K. E., Rosenberg, D. M. (1997). The reference condition: a comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. Journal of the North American Benthological Society, 16, 833-852.

Ricaurte, L.F., Boesch, S., Jokela, J., Tockner, K. (2012). The distribution and environmental state of vegetated islands within human-impacted European rivers. Freshwater Biology, 57(12), 2539-2549.

Richter, B.D., Postel, S., Revenga, C., Scudder, T., Lehner, B., Churchill, A., Chow, M. (2010). Lost in development's shadow: The downstream human consequences of dams. Water Alternatives, 3(2), 14-42

Riis, T., Olesen, B., Clayton, J.S., Lambertini, C., Brix, H., Sorrell, B.K. (2012). Growth and morphology in relation to temperature and light availability during the establishment of three invasive aquatic plant species. Aquatic Botany, 102, 56-64.

Rinaldi, M., Surian, N., Comiti, F., Bussettini, M., Nardi, L., Lastoria, B. (2015). IDRAIM: A Methodological Framework for Hydromorphological Analysis and Integrated River Management of Italian Streams. In Engineering Geology for Society and Territory-Volume 3 (pp. 301-304). Springer International Publishing.

Rockström, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J. (2009). A safe operating space for humans. Nature, 461, 472-475.

Rosenberg, D.M., Resh, V.H. (1993). Freshwater Bio monitoring and Benthic Macro invertebrates. Chapman and Hall, New York, US.

Rosenberg, D.M., Berkes, F., Bodaly, R. A., Hecky, R. E., Kelly, C. A., Rudd, J. W. (1997). Large-scale impacts of hydroelectric development. Environmental Reviews, 5(1), 27-54.

Rosenberg, D.M., McCully, P., Pringle, C.M. (2000). Global-scale environmental effects of hydrological alterations: introduction. BioScience 50(9), 746–751.

Rosenzweig, C., Casassa, G., Imeson, A., Karoly, D. J., Liu Chunzhen, Menzel, A., Rawlins, S., Root T. L., Seguin, B., Tryjanowski, P. (2007) Assessment of observed changes and responses in natural and managed systems. In: Climate Change 2007: Impacts, Adaptation and Vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (ed. by M. L. Parry, O. F. Canziani, J. P. Palutikof, P. J. van der Linden & C. E. Hanson), 79–131. Cambridge University Press, UK.

Rosenzweig, C., Karoly, D., Vicarelli, M., Neofotis, P., Wu, Q., Casassa, G. Menzel, A., Root, T.L., Estella, N., Seguin, B., Tryjanowski, P., Chunzhen, L., Rawlins, S., Imeson, A. (2008). Attributing physical and biological impacts to anthropogenic climate change. Nature, 453(7193), 353-357.

Rosgen, D. L. (1994). A classification of natural rivers. Catena, 22, 169-199.

Rosgen, D.L., Silvey, H.L. (1996). Applied River Morphology. Wildland Hydrology Books, Fort Collins, CO.

Rowan, J.S., Carwardine, J., Duck, R.W., Bragg, O.M., Black, A.R., Cutler, J.E.J., Soutar, I., Boon P.J. (2006). Development of a technique for Lake Habitat Survey (LHS) with applications for the European Union Water Framework Directive. Aquatic Conservation: Marine and Freshwater Ecosystems, 16,637-657.

RSA (Republic of South Africa) (1998). National Water Act . Act No. 36. Government Gazette, vol. 398, no. 19182. Cape Town.

Russian Federation Water Code No. 174-03 (2006). http://www.cabri-volga.org/DOC/PolicyRoundtable/WaterCodeOfRF-UnofficialEnglishTranslation.pdf [Accessed 02 February 2016]

Russi, D, ten Brink, P., Farmer, A., Badura, T., Coates, D., Förster, J., Kumar, R., Davidson, N. (2013). The economics of ecosystems and biodiversity for water and wetlands. Institute for European Environmental Policy. Ramsar Secretariat, Gland.

Ryding, S.-O., Rast, W. (1989). The control of eutrophication of lakes and reservoirs. Man and Biosphere Series, Volume 1, UNESCO and Parthenon Publishing Group, Paris.

SABESP (São Paulo State Sanitation Company) (2016). Agência de Notícias. http://www.sabesp.com.br/CalandraWeb/CalandraRedirect/?temp=4&proj=AgenciaNoticias&pub=T&db=&docid=681049976E1647F483257669005FF7A9 [Accessed 11 March 2016]

Saether, O.A. (1979). Chironomid communities as water quality indicators. Holarctic Ecology, 2, 65-74.

Saleth, R.M., Dinar, A. (2005). Water institutional reforms : theory and practice.Water Policy 7, pp.1–19; OECD, 2011. Water Governance in OECD Countries: A Multi-level Approach, OECD Publishing, Paris.

Sandin, L., Johnson, R.K. (2000). The statistical power of selected indicator metrics using macroinvertebrates for assessing acidification and eutrophication of running waters. Hydrobiologia, 422/423, 233-243.

Santore, R.C., Driscoll, C.T. (1995). The CHESS model for calculating chemical equilibria in soils and solutions. In: Loeppert, R., Schwab, A.P., Goldberg, S. (eds.), Chemical Equilibrium and Reaction Models. Soil Science Society of America Special Publication 42. American Society of Agronomy, Madison, WI, pp 357–375.

Santore, R.S., Di Toro, D.M., Paquin, P.R., Allen, H.E., Meyer, J.S. (2000). Biotic ligand model of the acute toxicity of metals. 2. application to acute copper toxicity in freshwater fish and daphia. Environmental Toxicology and Chemistry, 20, 2397-2402.

Schaumburg, J., Schranz, C., Foerster, J., Gutowski, A., Hofmann, G., Meilinger, P., Schneider, S. (2004). Ecological classification of macrophytes and phytobenthos for rivers in Germany according to the Water Framework Directive. Limnologica, 34 (4), 283-301.

SCHER (Scientific Committee on Healthand Environmental Risks), SCCS (Scientific Committee on Consumer Safety), SCENIHR (Scientific Committee on Emerging and Newly Identified Health Risks) (2012). Opinion on the Toxicity and Assessment of Chemical Mixtures, 50 p.

Scherman, P.-A., Muller, W.J., Palmer, C.G. (2003). Links between ecotoxicology, biomonitoring and water chemistry in the integration of water quality into environmental flow assessments. River Research and Applications, 19, 1–11.

Schindler, D. W., Smol, J. P. (2006). Cumulative effects of climate warming and other human activities on freshwaters of arctic and subarctic North America. AMBIO: A Journal of the Human Environment, 35(4), 160-168.

Schmeer, K. (1999). Guidelines for Conducting a Stakeholder Analysis.Partnerships for Health Reform, Abt Associates Inc., Bethesda, MD.

Schneider, S. C., Kahlert, M., Kelly, M. G. (2013). Interactions between pH and nutrients on benthic algae in streams and consequences for ecological status assessment and species richness patterns. Science of the Total Environment, 444, 73-84.

Schofield, N.J., Davies, P.E. (1996). Measuring the health of our rivers. Water-Melbourne Then Artarmon, 23, 39-43.

Scrimgeour, G.J., Wicklum, D. (1996). Aquatic Ecosystem Health and Integrity: Problems and Potential Solutions. Journal the North American Benthological Society, 15, 254-261.

Setiawan, F., Subehi, L., Wibowo, H., Matsushita, B. Fukushima, T. (2012). Preliminary study on remote sensing techniques to estimate water quality parameters at Lake Maninjau and Singkarak. Proceeding of National Seminar of Limnology. Cibinong 17 October. p 206- 225.

Settele, J., Scholes, R., Betts R., Bunn, S., Leadley, P., Nepstad, D., Overpeck, J.T., Taboada, M.A. (2014) Terrestrial and inland water systems. In: Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, and L.L. White (eds.). Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge and New York.

Shafique, N.A., Fulk, F., Autrey, B.C., Flotemersch, J. (2003). Hyperspectral remote sensing of water quality parameters for large rivers in the Ohio River basin. Proceedings First Interagency Conference on Research in the Watersheds, 216-221.

Shanahan, P., Harleman, D.R.F. (1986). Lake Eutrophication Model: A Coupled Hydrodynamic-Ecological Model. In: Somlyódy, L., and van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Sharpley, A.N., Chapra, S.C., Wedepohl, R., Sims, J.T. Daniel, T.C., Reddy, K.R. (1994). Managing agricultural phosphorus for the protection of surface waters: issues and opinions. Journal of Environmental Quality, 23, 437-451.

Sigel, K., Klauer, B., Pahl-Wostl, C. (2010). Conceptualising uncertainty in environmental decision-making: The example of the EU water framework directive. Ecological Economics, 69, 502-510.

Simpson, J.C., Norris, R.H. (1997). Biological assessment of water quality: development of AUSRIVAS models and outputs. In: Wright, J.F., Sutcliffe, D.W., Furse, M.T. (eds.). Assessing the biological quality of freshwaters: RIVPACS and similar techniques. Proceedings of an International Workshop held in Oxford, UK, on 16-18 September 1997, 2000pp, 125-142. Published by the Freshwater Biological Association, Ambleside.

Simpson, J. (2007). Safeguarding the World Water. Cabri-Volga Brief, 3/4. http://www.cabri-volga.org/DOC/CabriVolgaBriefIssue1.pdf [Accessed 02 February 2016]

Smith, B. (1995). On drawing lines on a map. In: Hirtle, S.C., Frank, A.U. (eds.). Spatial Information Theory A Theoretical Basis for GIS, 475-484. Springer Verlag, Berlin.

Smith, K.G., Barrios, V., Darwall, W.R.T., Numa, C. (eds.) (2014). The status and distribution of freshwater biodiversity in the Eastern Mediterranean. IUCN, Cambridge, Malaga and Gland.

Snelder, T.H., Biggs, B.J. (2002). Multiscale river environment classification for water resources management. Journal of the American Water Resources Association, 38(5), 1225-1239.

Snell, M., Irvine, K. (2012). Importance of scalar and riparian habitat effects for assessment of ecological status using littoral diatoms. Ecological Indicators, 2, 149-155.

Soininen, J., Bartels, P., Heino, J., Luoto, M., Hillebrand, H. (2015). Toward more integrated ecosystem research in aquatic and terrestrial environments. BioScience, biu216.

Solimini, A.G., Sandin, L. (2012). The importance of spatial variation of benthic invertebrates for the ecological assessment of European lakes. Fundamental and Applied Limnology, 180, 85-89.

Solomon, F. (2008). Impacts of Metals on Aquatic Ecosystems and Human Health. http://www.infomine.com/library/publications/docs/mining.com/Apr2008c.pdf [Accessed 18 February 2016]

Somlyódy, L. (1982). Modelling a Complex Environmental System: The Lake Balaton Study, Mathematical Modelling, 3(5), 481-502.

Somlyódy, L. (1986). Eutrophication Management Models. In: Somlyódy, L., van Straten, G. (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Somlyódy, L. and van Straten, G. (eds.) (1986). Modeling and managing shallow lake eutrophication – with application to Lake Balaton. Springer, Berlin.

Somlyódy, L., Jolánkai, G. (1986). Nutrient Loads. In: Somlyódy, L., van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Somlyódy, L., Pintér, J., Koncsos, L., Hanácsek, I., Juhász, I. (1986). Estimating averages and detecting trends in water quality data. IAHS-AISH, Publ. No. 157.

Somlyódy, L., Wets, R.J.-B. (1988). Stochastic Optimization Models for Lake Eutrophication Management. Operations Research, 36 (5).

Somlyódy, L., Clement, A., Istvánovics, V., G. Tóth, L., Jolánkai, G., Sisák, I., Padisák, J., Specziár, A. (2003). Evaluation of the state of water quality of Lake Balaton. Report. Department of Sanitary and Environmental Engineering, Budapest University of Technology. (In Hungarian).

Somlyódy, L., Honti, M. (2005). The Case of Lake Balaton: How can we exercise Precaution? Water Science Technology, 52 (6), 95-203.

Somlyódy, L. (2015a): Comparison of MAB, OECD and IWQGES classification systems. Institute of Sanitary Engineering, Technical University, Hungary.

Somlyódy, L. (2015b): Compilation of Balaton Lake basins TP loads – before and after introduction of load reduction measures. Institute of Sanitary Engineering, Technical University, Hungary.

Somvanshi, S., Kunwar, P., Singh, N.B, Shukla, S.P, Pathak, V. (2012). Integrated remote sensing and GIS approach for water quality analysis of Gomti river, Uttar Pradesh. International Journal of Environmental Sciences, 1(3), 62-74.

Søndergaard, M., Jensen, J.P., Jeppesen, E. (1999). Internal loading in shallow Danish Lakes. Hydrobiologia, 409, 145-152.

Soranno, P.A., Cheruvelil, K.S., Webster, K.E., Bremigan, M.T., Wagner, T., Stow, C.S. (2010). Using landscape limnology to classify freshwater ecosystems for multi-ecosystem management and conservation. BioScience, 60(6), 440-454.

Specziár, X. (1999). Nutrition strategy of five carp species at main habitats of Lake Balaton. Halászat 92. (In Hungarian).

Spikmans, F., van Tongeren, T., van Alen, T., van der Velde, G., den Camp, H. (2013). High prevalence of the parasite Sphaerothecum destruens in the invasive topmouth gudgeon Pseudorasbora parva in the Netherlands, a potential threat to native freshwater fish. Aquatic Invasions, 8(3), 355-360.

Stein, J.L., Stein, J.A., Nix, H.A. (2002). Spatial analysis of anthropogenic river disturbance at regional and continental scales: identifying the wild rivers of Australia. Landscape and Urban Planning 60, 1–25.

Stephen, C.E., Mount, D.I., Hansen, D.J., Gentile, J.H., Chapman, G.A., Brungs, W.A. (1985). Guidelines for deriving numeric national water quality criteria for the protection of aquatic organisms and their uses PB85 227049. U.S. Environmental Protection Agency, Washington, DC., USA.

Stevenson, J. (2014). Ecological assessments with algae: a review and synthesis. Journal of Phycology, 50, 437-461.

Steward, A.L., Tockner, K., Marshall, J.C., Schiller, D., Bunn, S.E. (2012). When the river runs dry: Human and ecological values of dry river beds. Frontiers in Ecology and Evolution, 10, 202–209.

Stoch, F., Galassi, D.M.P. (2010). Stygobiotic crustacean species richness: a question of numbers, a matter of scale. Hydrobiologia, 653 (1), 217 – 234.

Stoddard J.L., Larsen D.P., Hawkins C.P., Johnson R.K., Norris R.H. (2006). Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications, 16, 1267-1276.

Stoddard, J.L., Herlihy, A.T., Peck, D.V., Hughes, R.M., Whittier, T.R., Tarquinio, E. (2008). A process for creating multimetric indices for larger-scale aquatic surveys. Journal of the North American Benthological Society, 27, 878–891.

Stoffels, R.J., Clarke, K. R. and Closs, G. P. (2005). Spatial scale and benthic community organisation in the littoral zones of large oligotrophic lakes: potential for cross-scale interactions. Freshwater Biology, 50, 1131-1145.

Stokes, K., O'Neill, K., McDonald, R.A. (2004). Invasive species in Ireland. Report to Environment & Heritage Service and National Parks & W ildlife Service. Quercus, Queens University Belfast.

Straten, G. V. (1986). Identification, uncertainty assessment and prediction in lake eutrophication. Thesis, University of Twente, Enschede.

Strayer, D.L. (2006). Challenges for freshwater invertebrate conservation. Journal of the North American Benthological Society, 25, 271-287.

Strayer, D.L. (2009). Twenty years of zebra mussels: lessons from the mollusk that made headlines. Frontiers in Ecology and the Environment, 7, 135-141.

Strayer, D.L. (2010). Alien species in fresh waters: Ecological effects, interactions with other stressors, and prospects for the future. Freshwater Biology, 55(s1), 152-174.

Strayer, D.L., Dudgeon, D. (2010). Freshwater biodiversity conservation: recent progress and future challenges", Journal of the North American Benthological Society, 29, 344-358.

Stribling, J.B. (2011). Partitioning Error Sources for Quality Control and Comparability Analysis in Biological Monitoring and Assessment, Chapter 4. In: Eldin, A.B. (ed.). Modern Approaches to Quality Control. INTECH Open Access Publisher, 59-84.

Struijs, J., De Zwart, D., Posthuma, L., Leuven, R.S., Huijbregtsz, M.A. (2010). Field Sensitivity Distribution of Macroinvertebrates for Phosphorus in Inland Waters. Integrated Environmental Assessment and Management, 7(2), 280-286.

Sulastri, D., Hartoto Y., Yuniarti, I. (2012). Environmental Condition, Fish Resources and Management of Maninjau Lake of West Sumatera. Journal of Indonesian Fishery Research, 18 (1), 1 – 15.

Suter, G.W. II, Cormier, S. M. (2008). What is meant by risk-based environmental quality criteria? Integrative Environmental Assessment and Management, 4(4), 486-489.

Suter G.W. II, Norton, S.B., Cormier, S.M. (2010). The science and philosophy of a method for assessing environmental causes. Human and Ecological Risk assessment: An International Journal, 16(1), 19-34.

Suter, G.W. II, Cormier, S.M. (2011). Why and how to combine evidence in environmental assessments: Weighing evidence and building cases. Science of the Total Environment, 409, 1406-1417.

Suter, G.W. II, Cormier, S.M. (2013). A method for assessing the potential for confounding applied to ionic strength in central Appalachian streams. Environmental Toxicology and Chemistry, 32(2), 288–295.

Sweeney, B.W., Battle, J.M, Jackson, J.K. (2011). DNA barcodes of stream macroinvertebrates improve descriptions of community structure and water quality?. Journal of the North American Benthological Society, 30 (1), 195-216.

SWEPA (Swedish Environmental Protection Agency) (2000). Environmental Quality Criteria. Lakes and Watercourses. Swedish Environmental Protection Agency, Stockholm, 102 pp.

Syandry, H. (2000). Impact of cage aquaculture on water quality of Lake Maninjau, Presented in Panel Press Club (PPC), Padang. 13 pp.

Szilágyi, F., Somlyódy, L., Herodek, S., Istvánovics, V. (1989). The Kis-Balaton Reservoir System as Means of Controlling Eutrophication of Lake Balaton, Hungary. In: Jorgensen, S.E. (ed.). Management of Lakes and Reservoirs. Elsevier.

ter Braak C.J.F., Looman C.W.N. (1986). Weighted averaging, logistic regression and the Guassian response model. Vegetatio, 65, 3-11.

Tharme, R.E., King, J.M. (1998). Development of the B uilding B lock Methodolog y for In-stream Flow Assessments, and Supporting Research on the Effects of Different Magnitude Flows on Riverine

Ecosystems. Water Research Commission Report No. 576/1/98. Freshwater Research Unit, Cape Town.

Tharme, R.E. (2003). A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Research and Applications, 19, 397-441.

The Economist (2011). The Silvery Tietê – Cleaning up an open sewer. http://www.economist.com/node/21533415 [Accessed 02 February 2016]

Thienemann, A. (1931). Tropische Seen und Seetypen. Archiv für Hydrobiologie, Supplement, 9, 205-231.

Tipping, E., 1994. WHAM-a chemical equilibrium model and computer code for waters, sediments and soils incorporating a discrete site electrostatic model of ion binding by humic substances. Computers and Geosciences 20, 973–1023.

Tockner, K., Bunn, S.E., Gordon, C., Naiman, R.J., Quinn, G.P. and Stanford, J.A. (2008). Floodplains: Critically threatened ecosystems. In: N. Polunin (ed). Aquatic ecosystems. Trends and Global Prospects. Cambridge Press, 45-61.

Tolonen, K.T., Hämäläinen, H., Holopainen, I.J., Karjalainen, J. (2001). Influences of habitat type and environmental variables on littoral macroinvertebrate communities in a large lake system. Archiv für Hydrobiologie, 152, 39 - 67.

Townsend, C.R., Dolédec, S., Norris, R., Peacock, K., Arbuckle, C. (2003). The influence of scale and geography on relationships between stream community composition and landscape variables: description and prediction. Freshwater Biology, 48, 768–785.

Trenberth, K.E., Smith, L., Qian T., Dai, A., Fasullo, J. (2007). Estimates of the global water budget and its annual cycle using observational and model data. Journal of Hydrometeorology, 8, 758-769.

Turtle Conservation Fund (2002). A Global Action Plan for Conservation of Tortoises and Freshwater Turtles. Strategy and Funding Prospectus 2002–2007. Conservation International and Chelonian Research Foundation, Washington, DC.

UK TAG (UK Technical Advisory Group), Water Framework Directive) (2007). Recommendations on Surface Water Classification Schemes for the Purposes of the Water Framework Directive, UKTAG, pp. 61. http://www.wfduk.org/resources%20/recommendations-surface-water-classification-schemes. [Accessed 18 February 2016]

UK TAG (UK Technical Advisory Group, Water Framework Directive) (2008a). UK Environmental standards and conditions (Phase 1), Final report, April 2008. http://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Environmental%20standards%20phase%201_Finalv2_010408.pdf [Accessed 18 February 2016]

UK TAG (UK Technical Advisory Group, Water Framework Directive) (2008b). UK Environmental standards and conditions (Phase 2), Final report, March 2008. whttp://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Environmental%20standards%20phase%202_Final_110309.pdf [Accessed 18 February 2016]

UK TAG (UK Technical Advisory Group, Water Framework Directive) (2008c). Proposal for environmental quality standards for annex VIII substances, Final, January 2008 (revised June 2008). http://www.wfduk.org/sites/default/files/Media/Environmental%20standards/Specific%20pollutants%20proposals_Final_010608.pdf [Accessed 18 February 2016]

UK TAG (UK Technical Advisory Group) Water Framework Directive (2013). Final recommendations on new and updated standards. http://www.wfduk.org/sites/default/files/Media/UKTAG%20Final%20recommendations%20on%20biological%20stds_20131030.PDF [Accessed 02 February 2016]

UNECA (United Nations Economic Commission for Africa), EIS Africa (A Network for the Co-operative Management of Environmental and Geospatial Information in Africa), GSDI (Gloabl Spatial Data Infrastructure), ITC (International Institute for Geo-Information Science and Earth Observation) (2003). The SDI handbook for Africa; Chapter 5: Data Policy. United Nations Economic Commission for Africa, Development Information Services Division, Addis Ababa. http://repository.uneca.org/bitstream/handle/10855/5093/Bib-33547%20Add.5.pdf?sequence=8. [Accessed 18 February 2016]

UNECE (United Nations Economic Commission for Europe) (1994). Standard Statistical Classification of Surface Freshwater Quality for the Maintenance of Aquatic Life. In: Readings in International Environmental Statistics, New York and Geneva. http://unstats.un.org/unsd/envaccounting/ceea/archive/Framework/classification_in_environment.pdf [Accessed 09 March 2016]

UNGA (United Nations General Assembly) (2013). Draft decision submitted by the President of the General Assembly: Open Working Group of the General Assembly on Sustainable Development Goals, Sixty-seventh session, A/67/L.48/Rev.1, 15 January 2013

UNGA (United Nations General Assembly) (2015). Transforming our world: the 2030 Agenda for Sustainable Development. Resolution adopted by the General Assembly on 25 September 2015. http://www.un.org/ga/search/view_doc.asp?symbol=A/RES/70/1&Lang=E [Accessed 18 February 2016]

UNEP (United Nations Environment Programme) (2001). Asia – Pacific Environment Outlook 2. UNEP Regional Office Asia Pacific, Bangkok.

UNEP (United Nations Environment Programme) (in press). A Snapshot of the World's Water Quality: Towards global assessment assessments UNEP Report.

UNEP-DHI (UNEP-DHI Partnership – Centre on Water and Environment) (2016). Transboundary River Basins. A Global Comparative Assessment of Baseline Status and Future Trends. Volume III, River Basins.

UNESCAP (United Nations Social Commission for Asia and the Pacific) (2016). What is Good Governance? http://www.unescap.org/pdd/prs/projectactivities/ongoing/gg/governance.asp [Accessed 02 February 2016]

UNESCO (United Nations Educational, Scientific and Cultural Organization) (2013). Free Flow - Reaching Water Security Through Cooperation. UNESCO and Tudor Rose, Paris.

UNU-INWEH (United Nations University of Water, Environment and Health) and UNESCAP (United Nations Social Commission for Asia and the Pacific) (2013): Water Security & the Global Water Agenda. A UN Water Analytical Brief. United Nations University - Institute for Water, Environment & Health, 37 pages. http://www.unwater.org/downloads/watersecurity_analyticalbrief.pdf [Accessed 23 February 2016]

UN-Water (United Nations inter-agency mechanism on all freshwater related issues, including sanitation) (2011). Policy Brief: Water Quality. 19 pages. http://www.unwater.org/fileadmin/user_upload/unwater_new/docs/waterquality_policybrief.pdf [Accessed 23 February 2016]

UN-Water (United Nations inter-agency mechanism on all freshwater related issues, including sanitation) (2015). GEMI. Integrated monitoring of water and sanitation. http://www.unwater.org/fileadmin/user_upload/unwater_new/docs/GEMI_v2_April_2015.pdf [Accessed 23 February 2016]

US EPA (US Environmental Protection Agency) (1985). Guidelines for Deriving Numerical National Water Quality Criteria for the Protection Of Aquatic Organisms and Their Uses by C.E. Stephen, D.I. Mount, D.J. Hansen, J.R. Gentile, G.A. Chapman, and W.A. Brungs. Report PB85-227049. http://www.epa.gov/wqc/guidelines-deriving-numerical-national-water-quality-criteria-protection-aquatic-organisms-and [Accessed 02 February 2016]

US (US Environmental Protection Agency) EPA (1986). Quality Criteria for Water 1986 http://nptwaterresources.org/wp-content/uploads/2014/01/1986-goldbook.pdf [Accessed 18 February 2016]

US EPA (US Environmental Protection Agency) (1990). Biological Criteria: National Program Guidance for Surface Waters. http://nepis.epa.gov/Exe/ZyPDF.cgi/00001OKG.PDF?Dockey=00001OKG.PDF [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (1992). Procedure for Initiating Narrative Biological Criteria. http://nepis.epa.gov/Exe/ZyPDF.cgi/20003GN5.PDF?Dockey=20003GN5.PDF [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (1998) Guidelines for Ecological Risk Assessment. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC, EPA/630/R095/002F. http://www.epa.gov/sites/production/files/2014-11/documents/eco_risk_assessment1998.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2000a). Nutrient criteria technical guidance manual: Rivers and streams. EPA 822/B-00/002. Technical Report. Washington, DC. http://www.epa.gov/sites/production/files/documents/guidance_rivers.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2000b). Guidance for the data quality objectives process. EPA QA/G-4. EPA 600-R-96-055. Office of Environmental Information. Washington DC. https://www.orau.org/ptp/PTP%20Library/library/EPA/QA/g4.pdf [Accessed 23 February 2016]

US EPA (US Environmental Protection Agency) (2000c). Stressor Identification Guidance Document. EPA/822/B-00/025. Office of Research and Development and the Office of Water, Washington, DC, USA. https://clu-in.org/download/contaminantfocus/sediments/stressor-guidancej.pdf [Accessed 18 February 2016]

US EPA (US Environmental Protection Agency) (2000d). Nutrient Criteria Technical Guidance Manual. Lakes and Reservoirs. EPA-822-B-00-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

US EPA (US Environmental Protection Agency) (2002). Summary table for the Nutrient Criterial Documents. http://www2.epa.gov/sites/production/files/2014-08/documents/criteria-nutrient-ecoregions-sumtable.pdf [Accessed 18 February 2016]

US EPA (US Environmental Protection Agency) (2003). Generic Ecological Assessment Endpoints (GEAE) for Ecological Risk Assessment. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC, EPA/630/P-02/004F. http://www.epa.gov/sites/production/files/2014-11/documents/generic_endpoinsts_2004.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2005). Use of Biological Information to better define Designated Aquatic Life Uses in State and Tribal Water Quality Standards. EPA-822-R-05-001. Office of Water. http://nepis.epa.gov/Exe/ZyPDF.cgi/901I0600.PDF?Dockey=901I0600.PDF [Accessed 23 February 2016]

US EPA (US Environmental Protection Agency) (2006a). Framework for Developing Suspended and Bedded Sediment (SABS) Water Quality Criteria. U.S. Environmental Protection Agency, Washington, DC, EPA/822-R-06-001. http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=164423 [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2006b). Concepts and Approaches for the Bioassessment of Non-wadeable Streams and Rivers. EPA/600/R/06/127. http://nepis.epa.gov/Exe/ZyPDF.cgi/600006KV.PDF?Dockey=600006KV.PDF [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2007). Aquatic Live ambient Freshwater Quality Criteria-Copper. EPA-822-R-07-001. http://nepis.epa.gov/EPA/html/DLwait.htm?url=/Exe/ZyPDF.cgi/P1000PXC.PDF?Dockey=P1000PXC. PDF [Accessed 02 February 2016]

US EPA (Environmental Protection Agency) (2010a). Integrating Ecological Assessment and Decision-Making at EPA: A Path Forward. Results of a Colloquium in Response to Science Advisory Board and National Research Council Recommendations. Risk Assessment Forum. Washington, DC. EPA/100/R-10/004. http://www.epa.gov/sites/production/files/2013-09/documents/integrating-ecolog-assess-decision-making.pdf [Accessed 02 February 2016]

US EPA (Environmental Protection Agency) (2010b). Using stressor-response relationships to derive numeric nutrient criteria. EPA-820-S-10-001. http://www.epa.gov/sites/production/files/documents/using-stressor-response-relationships-to-derive-numeric-nutrient-criteria-pdf.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2010c). Causal Analysis/Diagnosis Decision Information System (CADDIS). Office of Research and Development, Washington, DV. http://www.epa.gov/caddis [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2011a). A Field-Based Aquatic Life Benchmark for Conductivity in Central Appalachian Streams (Final Report). U.S. Environmental Protection Agency, Washington, DC, EPA/600/R-10/023F. http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=233809 [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency). (2011b). A Primer on Using Biological Assessments to Support Water Quality Management, U.S. Environmental Protection Agency, Washington, DC, EPA 810-R-11-01) http://water.epa.gov/scitech/swguidance/standards/criteria/aqlife/biocriteria/upload/primer_update.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2011c). Review of field-based aquatic life benchmark for conductivity in central Appalachian streams. EPA SAB/11/006. Technical Report. Washington, DC.US

US EPA (US Environmental Protection Agency) (2012). An Optimization Approach to Evaluate the Role of Ecosystem Services in Chesapeake Bay Restoration Strategies. EPA/600/R-11/001.

US EPA (US Environmental Protection Agency) (2013a). Aquatic Live ambient Water Quality Criteria for Ammonia—Freshwater. EPA 822-R-13-001. http://www.epa.gov/sites/production/files/2015-08/documents/aquatic-life-ambient-water-quality-criteria-for-ammonia-freshwater-2013.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2013b). The bioassessment program review: Assessing level of technical rigor to support water quality management. EPA/ 820-R-13-001. Office of Water, 158 pp.

US EPA (US Environmental Protection Agency) (2013c) National Rivers and Streams Assessment 2008 -2009, draft report. http://water.epa.gov/type/rsl/monitoring/riverssurvey/upload/NRSA0809_Report_Final_508Compliant_130228.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2013d). A quick guide to developing Watershed Plans to restore and protect our Waters. EPA 841-R-13-003. http://water.epa.gov/polwaste/nps/upload/watershed_mgmnt_quick_guide.pdf [Accessed 02 February 2016

US EPA (US Environmental Protection Agency) (2014a). Water Quality Standards Handbook Chapter 3: Water Quality Criteria (40CFR 131.11). http://water.epa.gov/scitech/swguidance/standards/handbook/[Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2014b). External Peer Review Draft Aquatic Life Ambient Water Quality Criterion for Selenium – Freshwater, external peer review draft. EPA 822-P-14-001. U.S. Environmental Protection Agency, Washington, D.C. http://www.epa.gov/sites/production/files/2015-08/documents/external-peer-review-draft-aquatic-life-ambient-water-quality-criterion-for-selenium-freshwater-2014_0.pdf [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2015a). ECOTOX database http://cfpub.epa.gov/ecotox/ [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2015b). Summary of the Clean water Act. http://www2.epa.gov/laws-regulations/summary-clean-water-act [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2015c). Developing Total Maximum Daily Loads (TMDL) http://www.epa.gov/tmdl/developing-total-maximum-daily-loads-tmdl [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2016a). National Recommended Water Quality Criteria. http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm [Accessed 02 February 2016]

US EPA (US Environmental Protection Agency) (2016b). http://www.epa.gov/bpspill/water-benchmarks.html#gen2 [Accessed 02 February 2016]

USGS (US Geological Survey) (2013). National Climate Change & Wildlife Science Centers Data Sharing Policy. https://nccwsc.usgs.gov/content/data-policies-and-guidance. [Accessed 02 February 2016]

Van de Bund, Wouter. (2009). Water Framework Directive intercalibration technical report. Part 1: Rivers. Joint Research Centre Institute for Environment and Sustainability, European Commission. http://www.apambiente.pt/dqa/assets/intercalibration-2003-2007-technical-report---rivers.pdf [Accessed 19 February 2016]

Van der Schraaf, A.A.A. (2005). The compliance strategy of the Netherlands. Proceedings 7th International Conference on environmental compliance and enforcement, 2005, Marrakech, Morocco, Volume 1. http://www.inece.org/conference/7/vol1/15_Vand%20Der%20Schraaf.pdf [Accessed 02 February 2016]

van Liere, L., Gulati, R.D. (eds.) (1992). Restoration and Recovery of Shallow Eutrophic Lake Ecosystems in the Netherlands. Developments in Hydrobiologia, 74. Kluwer Academic Publishers, 305 pp..

Van Sickle, J. (2010). Correlated metrics yield multimetric indices with inferior performance. Transactions of the American Fisheries Society 139.6, 1802-1817.

van Straten, G. (1986a). Lake Eutrophication Models. In: Somlyódy, L., van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

Van Straten, G. (1986b). Hypothesis Testing and Parameter Uncertainty Analysis in Simple Phytoplanton-P Models. In: Somlyódy, L., van Straten, G., (eds.). Modeling and Managing Shallow Lake Eutrophication. With application to Lake Balaton. Springer Verlag, Berlin.

van Vliet, M.T.H., Yearsley, J. R., Ludwig, F., Vogele, S., Lettenmaier, D.P., Kabat, P. (2012). Vulnerability of US and European electricity supply to climate change. Nature Climate Change, 2, 676–81.

Vaughan, I.P., Diamond, M., Gurnell, A.M., Hall, K.A., Jenkins, A., Milner, N.J., Naylor, L.A., Sear, D.A., Woodward, G., Ormerod, S.J. (2009). Integrating ecology with hydromorphology: A priority for river science and management. Aquatic Conservation: Marine and Freshwater Ecosystems, 19, 113–125.

Veiga, L. B. E., Magrini, A. (2013). The Brazilian water resources management policy: fifteen years of success and challenges, Water Resources Management, 27, 2287–2302.

Verdonschot, P. F. (2006). Evaluation of the use of Water Framework Directive typology descriptors, reference sites and spatial scale in macroinvertebrate stream typology. In: Furse, M.T., Hering, D., Brabec, K., Buffagni, A., Sandin, L., Verdonschot, P. (eds.). The Ecological Status of European Rivers: Evaluation and Intercalibration of Assessment Methods. Springer Verlag, 39-58.

Vogel, R. (2011). "Hydromorphology." Journal of Water Resources Planning and Management, 137(2), 147-149.

Vollenweider, R.A., Kerekes, J. (1982). Eutrophication of waters . Monitoring assessment and Control. Organization for Economic Co-operation and Development (OECD), Paris.

von der Ohe, P. C. & Liess, M. (2004). Relative sensitivity distribution of aquatic invertebrates to organic and metal compounds. Environmental Toxicology and Chemistry, 23(1), pp. 150-156.

von Schiller, D., Martí, E., Riera, J.L., Ribot, M., Marks, J.C., Sabater, F. (2008). Influence of land use on stream ecosystem function in a Mediterranean catchment. Freshwater Biology, 53, 2600–2612.

von Schiller D, Acuña V, Graeber D, Martí E, Ribot M, Sabater S, Timoner X, Tockner K (2011). Contraction, fragmentation and expansion dynamics determine nutrient availability in a Mediterranean forest stream. Aquatic Sciences 73, 485–497.

Vörösmarty, C.J., Sahagian, D. (2000). Anthropogenic disturbance of the terrestrial water cycle. BioScience, 50(9), 753-765.

Vörösmarty, C.J., Meybeck, M., Fekete, B., Sharma, K., Green, P., Syvitski, J. P. (2003). Anthropogenic sediment retention: Major global impact from registered river impoundments. Global and Planetary Change, 39(1), 169-190.

Vörösmarty, C.J., Lettenmaier, D., Leveque, C., Meybeck, M., Pahl-Wostl, C., Alcamo, J., Cosgrove, W., Grassl, H., Hoff, H., Kabat, P., Lansigan, F., Lawford, R., Naiman, R. (2004). Humans transforming the global water system. Eos, Transactions American Geophysical Union, 85(48), 509-514.

Vörösmarty, C.J., McIntyre, P. B, Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Reidy Liermann C., Davies, P. M. (2010). Global threats to human water security and river biodiversity. Nature, 467, 555-561.

Vörösmarty, C.J. (2013). Humans in the Anthropocene: Masters of Integrated Water Management or Bulls in the China Shop?. Keynote lecture, Budapest Water Summit, 9th October 2013. Budapest, Hungary. www.budapestwatersummit.hu/budapest-water-summit/summit-plenavy/

Vos, P., Meelis, E., ter Keurs, W.J. (2000). A framework for the design of ecological monitoring programs as a tool for environmental and nature management. Environmental Monitoring and Assessment, 61, 317–344.

Walczykowski, P., Jenerowicz, P., Orych, A. (2013). A review on remote sensing methods of detecting physical water pollutants. In: Proceeding of research conference in technical disciplines. At Slovakia, 1, 18-22.

Walker, B., Holling, C.S., Carpenter, S.R., Kinzig, A. (2004). Resilience, adaptability and transformability in social–ecological systems. Ecology and Society 9(2), 5.

Walker, B., Meyers, J.A. (2004). Thresholds in ecological and social – ecological systems: a developing database. Ecology and Society, 9(2), 3.

Wallace, J.S., Acreman, M.C. and Sullivan, C.A. (2003). The sharing of water between society and ecosystems: from conflict to catchment-based co-management. Philosophical Transactions of the Royal Society of London Series B-Biological Sciences 358, 2011-2026.

Walley, W.J., Fontama, V. N., (1998a). Neural network predictors of average score per taxon and number of families at unpolluted river sites in Great Britain. Water Research, 32, 613-622.

Walley, W.J., Fontama, V. N., (1998b). Authors' response to comments on 'Neural network predictors of average score per taxon and number of families at unpolluted river sites in Great Britain'. Water Research, 32, 3502-3503.

Walters C.J. (1986). Adaptive Management of Renewable Resources. McGraw-Hill, New York.

Walters, C. (1997). Challenges in adaptive management of riparian and coastal ecosystems. Conservation Ecology, 1, 1.

Ward, J.V., Tockner, K., Edwards, P.J., Kollmann, J., Gurnell, A.M., Petts, G.E., Bretschko, G., Rossaro, B. (2000). Potential role of island dynamics in river ecosystems. Verhandlungen des Internationalen Verein Limnologie, 27, 2582-2585.

Warne, M.St.J. (2001). Derivation of the Australian and New Zealand Water Quality Guidelines for Toxicants. Austr. Journal of Toxicology, 7. 123-136.

Warner J. (ed.) (2007), Multi-Stakeholder Platforms for Integrated Water Management, Ashgate Publishing Ltd., Hampshire.

Warner, J., Wester, P., Bolding, A. (2008). Going with the flow: river basins as the natural units for water management? Water Policy, 10(S2), p.121.

Warner, A.T., Bach L.B., Hickey J.T. (2014). Restoring environmental flows through adaptive reservoir management: planning, science, and implementation through the Sustainable Rivers Project. Hydrological Sciences Journal 59(3-4): 770-785.

Wasson, J.B., Chandesris, A., Pella, H., Blanc, L. (2002). Typology and reference conditions for surface bodies in Franch –the hydro-ecoregion approach.Typology classification of lakes and rivers. Finnish Environment Institute (SYKE), Helsinki, Findland, October 25-26. TemaNord. Water Science Technology 52 (6), 95-203.

WCD (World Commission on Dams) (2000). Dams and Development. A new framework for decision-making. Earthscan Publications Ltd, London and Skilling, VA.

WEPA (Water Environment Partnership in Asia) (2009). Outlook of water environmental management strategies in Asia. Ministry of the Environment, Japan, IGES.

WEPA (Water Environment Partnershipin Asia) (2012). Outlook of water environmental management strategies in Asia. Ministry of the Environment, Japan, IGES.

Westervelt, J. (2001). Simulation modeling for watershed management. Springer Verlag, Berlin.

Wetzel, R.G. (2001). Limnology. Lake and River Ecosystems. Academic Press, San Diego.

White, J., Irvine, K. (2003). The use of littoral mesohabitats and their macroinvertebrate assemblages in the ecological assessment of lakes. Aquatic Conservation: Marine and Freshwater Ecosystems, 13, 331-351.

WHO (World Health Organization)(2011). Guidelines for drinking-water quality. WHO Press, Geneva.. http://apps.who.int/iris/bitstream/10665/44584/1/9789241548151_eng.pdf [Accessed 10 March 2016]

Wiederholm, T., Johnson R.K. (1997). Monitoring and Assessment of Lakes and Watercourses in Sweden, Temporal Reference Rivers, Department of Environmental Assessment, Swedish University for Agricultural Sciences.

Willby, N.J. (2011). From metrics to Monet: the need for an ecologically meaningful guiding image. Aquatic Conservation: Marine and Freshwater Ecosystems, 21, 601-603.

Williams, B.S., D'Amico, E., Kastens, J.H., Thorp, J.H., Flotemersch, J.E., Thoms, M.C. (2013). Automated riverine landscape characterization: GIS-based tools for watershed- scale research, assessment, and management. Environmental Monitoring and Assessment. Environmental Monitoring and Assessment, 185(9), 7485-7499.

Winder, M., Schindler, D. E. (2004). Climatic effects on the phenology of lake processes. Global Change Biology, 10, 1844–1856.

Wolf, A. T., Natharius, J. A., Danielson, J. J., Ward, B. S., Pender, J. K. (1999). International river basins of the world. International Journal of Water Resources Development, 15(4), 387-427.

Wolfram, G., Kowarc, V.A., Humpesch, U.H., Siegl, W. (2002). Distribution pattern of benthic invertebrate communities in Traunsee (Austria) in relation to industrial tailings and trophy. Water, Air, and Soil Pollution, 2, 63-91.

Wolter. C, Lorenz, S., Scheunig, S., Lehmann, N., Schomaker, C., Nastase, A., García de Jalón, D., Marzin, A., Lorenz, A., Kraková, M., Brabec, K., Noble, R. (2013). Review on ecological response to hydromorphological degradation and restoration. REFORM (REstoring rivers FOR effective catchment Management) Project. http://www.reformrivers.eu/system/files/1.3%20Ecol%20response%20to%20HyMo%20changes_final.pdf [Accessed 02 February 2016]

Wright, J.F. (1995). Development and use of a system for predicting macroinvertebrates in flowing waters. Australian Journal of Ecology, 20, 181–197.

Wright, J.F. (2000). An introduction to RIVPACS. In: Wright J.F., Sutcliffe D.W., Furse M.T. (eds.) Assessing the Biological Quality of Freshwaters. RIVPACS and other techniques. Freshwater Biological Association, 1 – 23.

Wright, D.A., Welbourn, P. (2002). Environmental Toxicology. Cambridge environmental chemistry series,11. Cambridge University Press, New York.

WWF (World Wildlife Fund) (2014.) HydroSHEDS. (2014). http://www.hydrosheds.org [Accessed 16 February 2016]

WWF (World Wildlife Fund) (2016). Global Lakes and Wetlands Database. http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database [Accessed 16 February 2016]

Xenopoulos, M. A., Lodge, D. M., Alcamo, J., Märker, M., Schulze, K., Van Vuuren, D. P. (2005). Scenarios of freshwater fish extinctions from climate change and water withdrawal. Global Change Biology, 11(10), 1557-1564.

Yamazaki, K. (2011). Regulatory Standards for Conservation of Aquatic Life in Japan. http://www.biosch.hku.hk/eqspae/speakers.html [Accessed 02 February 2016]

Yoder, C.O., Rankin, E.T. (1995). Biological response signatures and the area of degradation value: new tools for interpreting multimetric data. In: Davis, W.S., Simon, T.P. (eds). Biological assessment and criteria. Tools for water resource planning and decision making. Lewis Publishers, 263-286.

Yoder, C.O.,Rankin, E.T (1996). Assessing the condition and status of aquatic life designated uses in urban and suburban watersheds, pp. 201-227. in Roesner, L.A. (ed.). Effects of Watershed Development and Management on Aquatic Ecosystems, American Society of Civil Engineers, New York, NY. http://www.epa.state.oh.us/portals/35/documents/watrshed.pdf [Accessed 02 February 2016]

Zarfl, C., Lumsdon, A.E., Berlekamp, J., Tydecks, L., Tockner, K. (2015). A global boom in hydropower dam construction. Aquatic Sciences, 77, 161-170

Zhang, K.M., Wen, Z.G. (2008). Review and challenges of policies of environmental protection and sustainable development in China. Journal of Environmental Management, 88(4), 1249-1261.

Zhang, Y., Arthington, A.H., Bunn, S.E., MacKay, S., Xia, J., Kennard, M. (2012). Classification of flow regimes for environmental flow assessment in regulated rivers: The Huai River Basin, China. River Research and Applications, 2, 989-1005.

Zhen-guang, Y., Hong, W. Yi-zhe, W., Ya-hui, Z. Ruo-zhen, Y.,Jun-li, Z., Leung, K.M.Y., Zheng-tao, L. (2013). Developing a national water quality criteria system in China. Water Policy 15 (6), 936-942.

Zhou, Huai-dong, Cui-ling Jiang, Li-qin Zhu, Xin-wei Wang, Xiao-qin Hu, Jun-yu Cheng and Ming-hua Xie. (2011). Impact of pond and fence aquaculture on reservoir environment Water Science and Engineering, 4(1), 92-100.

Annex 2

Resolution of the UNEP Governing Council GC 27/3

The Governing Council,

Recalling its decision SS/XII/6 of 22 February 2012 on the world environmental situation, where it is recognized that there are gaps in our knowledge of the state of the environment resulting from a lack of data and regular monitoring, particularly in areas such as freshwater quality and quantity, estuarine and ocean water quality, groundwater depletion, ecosystem services, loss of natural habitat, land degradation and chemicals and wastes,

Recalling its decision 26/14 of 24 February 2011 to revitalize the Global Environment Monitoring System/Water Programme (GEMS/Water),

Recalling also the Washington Declaration on the Protection of the Marine Environment from Land-based Activities, and the Manila Declaration of the Third Intergovernmental Review Meeting of the implementation of the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA), which identifies nutrients, litter and waste water management as GPA priority areas,

Recalling that 2005–2015 is the United Nations International Decade for Action, “Water for Life”, and that 2013 is the United Nations International Year of Water Cooperation,

Reaffirming the role of the United Nations Environment Programme as the leading global environmental authority and principal body within the United Nations system in the field of environment, including global water quality monitoring and assessment,
Recalling further paragraphs 122 and 124 of the Outcome document of the United Nations Conference on Sustainable Development (Rio + 20), “The future we want”, which stresses the need to adopt measures to significantly reduce water pollution and increase water quality and recognizes the key role that ecosystems play in maintaining water quantity and quality,

Recognizing the availability of international guidelines for drinking water quality, for agriculture and drainage, for waste water reuse and other similar matters and the absence of international water quality guidelines for ecosystems,
Noting that water is essential for human life, the environment and the economy and thus action to protect water resources and promote its sustainable use is essential to the achievement of sustainable development,

Recognizing that water is at the core of sustainable development and is closely linked to a number of key global challenges and reiterating therefore the importance of integrating water into sustainable development and underlining the critical importance of water and sanitation within the three dimensions of sustainable development as referred to in paragraph 119 of “The future we want”,

Noting the alarming trends in water quality degradation and its negative impact on ecosystems functioning and human wellbeing and development,
Noting targets 8, 11 and 14 of the Aichi Biodiversity Targets under the Convention on

Biological Diversity, regarding water pollution control for ecosystem conservation,

Recognizing that there is a need for international water quality guidelines, which may be voluntarily used by Governments to maintain and improve the status of ecosystems to sustain the services they provide, as a possible basis for managing water pollution and water quality, as they affect ecosystems, and to support decision-making,
20

Recognizing also that water quality guidelines for ecosystems should be coherent and integrated, as appropriate, into existing guidelines related to water to promote its sustainable management,

1. Requests the Executive Director, in partnership with Governments, scientific institutions, United Nations agencies and other relevant stakeholders, particularly those from developing countries, to develop international water quality guidelines for ecosystems that may be voluntarily used to support the development of national standards, policies and frameworks, taking into account existing information while integrating, as appropriate, all relevant aspects of water management;
2. Encourages Governments, scientific institutions, United Nations agencies and other relevant organizations, including in the private sector, to participate actively in developing the water quality guidelines consistent with the preceding paragraph;
3. Invites Governments and others in a position to do so, including the private sector, to technically and financially support the process for the development of international water quality guidelines consistent with paragraph 1 of the present decision;
4. Requests the Executive Director to report to the Governing Council/Global Ministerial Environment Forum at its twenty-eighth session on the implementation of the present decision.

Annex 3

Physical and chemical criteria to assess quality of freshwater ecosystems and proposed benchmark values

In this annex benchmark values for physical and chemical indicators of freshwater ecosystem quality status are proposed which are indicative of high ecosystem integrity and extreme impairment, respectively. The first benchmark value will separate freshwater ecosystems of high integrity (Category 1) from ecosystems in worse quality status. The second benchmark demarcates the lower end of the quality continuum. Ecosystem quality status should be above this threshold, otherwise the water body would lose aquatic diversity and beneficial use and ecosystems will face severe reduction or complete loss of Ecosystem Services (ES)s (Category 4). The categories that characterize freshwater ecosystem quality are stated in »Figure 2.2. Values to demarcate minimally to moderate disturbed ecosystems status (Category 2) from highly disturbed ecosystems (Category 3) represent intermediate thresholds. The determination of such intermediate thresholds is however subject to specific considerations because of local physical, chemical and morphological conditions and local interpretation and is therefore not covered in detail in this document.

Benchmark values are presented for the

- oxygen regime
- nutrients (phosphorus and nitrogen) and Chlorophyll a (Chl-a)
- ammonia
- pH
- temperature
- heavy metals (aluminium, arsenic, cadmium, chromium, copper, lead, mercury, nickel and zinc).

The proposed benchmark values are based on internationally and nationally established criteria to protect aquatic life and to characterize severe ecosystems degradation (see »Table 4.8 for complete list). The selection of countries and entities for comparison of Water Quality Criteria (WQC) and Standards (WQS) in this annex is based on the overview of existing Water Quality Guidelines (WQGs) provided in »Chapter 3. The physical and chemical WQC and WQS considered for comparison stem from the following countries and entities:

- Australia/New Zealand
- Canada
- China
- European Union (for numerical values the United Kingdom standards are used, with the exception of priority pollutants)
- Japan,
- South Africa,
- United Nations Economic Commission for Europe (UNECE)
- US.

In the European Union Environmental Quality Standards (EQSs) are established at the community level for a limited number of priority pollutants. For all other indicators, the establishing of numerical quality standards to underpin the implementation of the Water Framework Directive (EU WFD) is the responsibility of the Member States. If standards at the community level are not available, the UK standards are used in this annex for comparison with guideline values in other countries and entities. For each group of indicators the ecological relevance, natural variations and regional differences are briefly indicated. For simplicity, benchmark values are expressed as annual average concentrations, unless indicated otherwise; noting also that many countries apply a more advanced monitoring strategy taking for example into account in-situ variations in time and place.

Selection of values

In the selected WQGs mutually different terminology is used and mutually different levels of protection are defined. It should be emphasized that the values presented in the tables in this annex may not always provide the same level of protection (Category 1) or the same level of impairment (Category 4). A short summary of the defined levels of protection in the guidelines considered is stated in this section. The next sections go in more detail concerning the numerical criteria and standards that are established for each group of indicators in the selected guidelines.

In Australia and New Zealand guidelines (ANZECC/ARMCANZ, 2000a) the term “Guideline trigger value” (and in the past “default value”) is used. Trigger values are concentrations that, if exceeded, would indicate potential environmental problems, and so “trigger” a management response, e.g. further investigations and subsequent refinement of the guidelines according to local conditions. For a large number of toxicants trigger values are derived on different level of protection: 99%, 95%, 90% and 80% of the species, using a statistical distribution approach based on single-species toxicity tests on a range of test species such as algae, daphnia and certain fish species. The rationale and background information for each indicator is presented in Volume 2 of the guidelines (ANZECC/ARMCANZ, 2000b). The 99% level of protection is used in this annex for comparison of values for high integrity in different guidelines.

The Canadian WQGs and subsequent updates (CCME, 2014a) set national science-based goals for the quality of aquatic ecosystems. Numerical guidelines for chemical substances are derived according to a general protocol, both for long-term exposure and short-term exposure (CCME, 2007). The long-term exposure values identify benchmarks (maximum concentrations) that are intended to protect all forms of aquatic life (all species, all life stages) for indefinite exposure times. The short-term exposure identifies benchmarks that protect only a specified fraction of individuals from severe effects such as lethality. For a large number of pollutants factsheets are available in which the derivation of the guideline values is described (CCME, 2014b). Long-term values are used to compare values for high integrity and short-term values for extreme impairment.
The state of the surface water quality in China is expressed in a range from Grade I to V. Physical and chemical WQS are established for each grade. The highest quality class (Grade I) is mainly for headstream and the national nature preserves. Grade II refers to drinking water resources in first class protected areas, and protected areas for precious fish, and spawning areas for fish and shrimp. The lowest class (Grade V) refers to water which is mainly used for agricultural water resources. Standards are derived for general physical and chemical indicators and a number of heavy metals (China, 2002). The standards for Grade I are used for the comparison with guideline values for high integrity from other countries and the standards for Grade V are used for the comparison of values for extreme impairment.

In the EU WFD (EC, 2009) five classes are distinguished in order to assess ecological status: high, moderate, good, poor and bad status. A high status means that there are no, or only very minor, anthropogenic alterations. It is assumed that the levels to achieve a certain status are type-specific, i.e. depending on the type of water body and its environmental context. All Member States shall derive general Water Quality Standards (WQSs). The available standards for the high and the bad status in the UK are used in this annex. Concerning toxicants two classes are distinguished: “good” or “failing” to achieve good status. The method for deriving EQSs for priority pollutants is established in the EUWFD Guidance document (EC, 2011b). The standards for a good status are used for the comparison with guideline values for high integrity from other countries and entities.

In Japan EQSs are set for the conservation of the living environment (MoEJ, 1997). The standards for class AA are established to protect all kinds of uses and the conservation of natural environment. These standards are used for the comparison of guideline values for high integrity from other countries and entities. The lowest class E is related to restricted conservation of the environment: limit or not disrupting the day-to-day lives of the population. If the quality standard is below the boundary for the lowest class E the value is supposed to be indicative for extreme impairment.

In South Africa Target Water Quality Ranges (TWQRs) for aquatic ecosystems are established (DWAF, 1996a). A TWQR gives the range of concentrations or levels within which no measurable adverse effects are expected on the health of aquatic ecosystems, and therefore ensure their protection. The TWQR is not a Water Quality Criterion (WQC), but rather a management objective which has been derived from quantitative and qualitative criteria. A TWQR is based on studies concerning Chronic Effect Values (CEVs), but a TWQR is below the CEV because a certain safety factor is applied. The CEV is defined as the concentration or level of a constituent at which there is expected to be a significant probability of measurable chronic effects to up to 5% of the species. The guideline also provides Acute Effect Values (AEVs). The AEV is defined as that concentration above which there is expected to be a significant probability of acute toxic effects to up to 5% of the species. The AEV is a criterion used to identify those cases requiring urgent management attention because the aquatic environment is threatened, even if the situation persists only for a brief period. The TWQRs are used for the comparison with guideline values for high integrity from other countries and the AEV values are used for the comparison of values for extreme impairment.

The UNECE developed a classification of surface water quality for the maintenance of aquatic life (UNECE, 1994). Class I refers to high quality systems without anthropogenic pollution. Class V means the quality of the surface water is very poor, e.g. acute problems in the oxygen regime, hypertrophic systems or acute toxic conditions in terms of concentration levels, duration and frequency, prevail. For each group of indicators the class interpretation is specified.

The United States Environmental Protection Agency (US EPA) provides national recommended WQC: narrative or numerical WQC for general indicators and pollutants in the US (US EPA, 2016). Objectives and standards can be set by states and tribes. The recommended WQC for aquatic life in the US include a list of approximately 60 substances most of them are toxic pollutants. The criteria contain two expressions of allowable magnitude: (i) Criterion Maximum Concentration (CMC): the highest concentration of a material to which an aquatic community can be exposed briefly without resulting in an unacceptable effect. (ii) Criterion Continuous Concentration (CCC): the highest concentration of a material to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect. In the year 1985 guidelines for deriving these criteria have been published which are available on the website of the US EPA (US EPA, 2016a). US EPA intends to consider information regarding new and alternative methods for deriving aquatic life criteria in order to inform revisions of US EPA's existing guidance documents using the newest, most appropriate science available.

Rationale

As all criteria and standards are the result of extensive and often long-lasting scientific, public and political debates in the considered countries or regions, the established levels of protections are not discussed in this annex and are furthermore not evaluated in relation to standards and criteria in other countries and regions. All criteria and standards in existing guidelines are compared and benchmarks are proposed. If there is a large range in the considered values, the benchmarks proposed in this document are close to the median values of the criteria and standards in the guidelines considered.
The proposed benchmarks may differ from the WHO drinking-water guidelines (WHO, 2011). As the preventive management of drinking-water quality and the protection of aquatic life both are important aspects of integrated water resource management, the proposed benchmarks may be relevant for the protection of drinking-water sources. For that reason, the WHO drinking-water guideline values for nitrate, nitrite and heavy metals are compared with the proposed benchmarks. For the other indicators referred to in this annex, WHO drinking-water guideline values were not established.

Oxygen regime

Multicellular organisms like fish and invertebrates, and many single-celled organisms like bacteria require oxygen for respiration, although some specialised forms can survive at very low oxygen concentrations because of biochemical or behavioural adaptations. However, the minimum level of Dissolved Oxygen (DO) to maintain higher forms of life varies strongly. Most fish-species require reasonably high levels of oxygen, whereas bottom feeders, crabs, mussels and worms can survive for long periods at rather low levels DO.

The percentage of DO saturation (DO %) is the amount of oxygen relative to the total amount of oxygen that the water can hold in equilibrium at a given temperature. 100 % air saturation means that the water is holding as many dissolved gas molecules as it can in equilibrium with the gas in the atmosphere. Supersaturation (>100%) may occur owing to the process of photosynthesis, evident in many nutrient enriched lakes or dense weed beds, or by rapid aeration as of frequent occurrence beside hydro-power dams, large waterfalls or highly turbulent rivers. Values below 100% saturation may be caused by microbiological decomposition of organic material from natural sources or discharges of organic waste, and other oxygen demanding chemicals, such as ammonium. In the lower layer of stratified lakes (i.e. in the hypolimnion) low saturation of DO is quite usual, because of lack of oxygen input and because the rate of oxygen used for decomposition of organic material exceeds replenishment of oxygen for the upper layer of the water column. Low DO saturation levels might be even accentuated in nutrient enriched lakes, but can also occur under low nutrient conditions where stratification of water occurs over long periods of time, or is permanent.

DO concentrations vary depending on temperature, pressure and salinity, e.g. at 4^0C DO concentrations are theoretically maximal at 13.09 mg/l, at 20^0C the maximum is at 9.07 mg/l. Benchmarks for oxygen concentrations are best derived from local conditions of temperature, pressure and salinity. Oxygen concentrations are constantly affected by diffusion, aeration, photosynthesis, respiration and decomposition resulting in daily, seasonal, vertical and horizontal fluctuations. Therefore, monitoring programs for oxygen should take into account such fluctuations.

Biological Oxygen Demand (BOD) measures the amount of oxygen consumed by microorganisms, mostly measured in a 5-day lab experiment (BOD_5). The greater the oxygen demand, the more rapidly oxygen will be depleted in the water. If the oxygen concentration is low and/or strongly fluctuating, measuring BOD_5 may provide important information in addition to DO measurements. Instead of BOD_5, Chemical Oxygen Demand (COD) can also be measured. However it is preferable to use DO or DO% as benchmark for indication of the quality for aquatic life, as the link between BOD_5 and COD is not always clear.

An excellent, easy to read and detailed explanation about the relevance of oxygen and fluctuations in oxygen concentrations in aquatic systems is given by Kemker (2013b). This publication includes a table for calculating DO concentrations at 100% air saturation for different temperatures and levels of salinity. The Australian and New Zealand guidelines (ANZECC/ARMCANZ, 2000b) and the South African Guidelines (DWAF, 1996a) present valuable background information on oxygen concentrations in water.

Table A 1: Overview of some Water Quality Criteria (WQC) for Dissolved Oxygen Saturation, Dissolved Oxygen Concentration and BOD_5 for high ecosystem integrity in freshwaters
[1] DO-values between brackets apply to early life stages
[2] Value of 8.0 mg/l for cyprinid fish species and 9.0 for salmonid species in lakes

	DO Saturation (%)	DO Concentration (mg/l)	BOD_5 mg/l	Reference
Australia/ New Zealand	80 – 120	-	-	ANZECC/ARMCANZ (2000a)
Canada[1] Warm water Cold water		5.5 (6.0) 6.5 (9.5)		CCME (1999)
China		7.5	≤ 3	China (2002)
EU - UK	> 80	8.0 -9.0 [2]	3	UKTAG (2008a)
Japan		7.5	1	MoEJ (1997)
South Africa	80 – 120			DWAF (1996a,b)
UNECE	90 – 110	7	-	UNECE (1994)
US[1] Warm water Cold water		5.5 (6.0) 6.5 (9.5)		US EPA (1986)

Existing quality criteria and standards for the oxygen regime (extreme impairment)
Both in China and Japan, standards for the lowest quality class are established for oxygen concentration and BOD_5.

Based on the EU WFD, in the UK standards for poor DO% are derived. The values (90-percentile) are < 50% for upland and low alkalinity rivers and < 45% for highland and high alkalinity rivers. This corresponds with a value of about 4 mg/l. The BOD standards are less than 7.5 mg/l and 9 mg/l, respectively. In Japan the minimum quality standard for DO has been established at ≥ 2 mg/l and for BOD ≤ 10 mg/l, based on daily averages.
In the South Africa WQGs minimum allowable values for oxygen saturation are given. The minimum sub lethal value is > 60% (7-day minimum) and the minimum lethal value is > 40% (1-day minimum). Both values should be applied together.

The boundaries for the lowest class of surface water quality published by UNECE (1994) are DO % < 30% and >150% for the upper layer of stratified waters and < 10% and > 150% for unstratified waters. The presented criterion for DO concentration is 3 mg/l and for COD > 30mg/l.

The US EPA criterion for 1 day minima DO concentrations is 5.0 mg/l for the early life stages and 3.0 mg/l for other life stages in warm water, and 5.0 mg/l and 5.5 mg/l, respectively in cold water. The criteria for cold water fish are intended to apply to waters containing a population of one or more species of the family of Salmonidia. The warm water criteria are intended to protect early life stages of warm water fish as channel catfish and largemouth bass.

An overview of WQC and WQS for bad conditions of the oxygen regime in a number countries and entities are presented in »Table A 2.

Table A 2: Overview of some Water Quality Criteria (WQC) for Dissolved Oxygen (DO) Saturation, DO Concentration and BOD_5 (Biochemical Oxygen Demand) for extreme impairment in freshwater ecosystems.

	DO Saturation (%)	DO Concentration (mg/l)	BOD_5 (mg/l)	Reference
Australia/ New Zealand	-	-	-	ANZECC/ARMCANZ (2000a)
Canada	-	-	-	CCME (1999)
China		3	10	China (2002)
EU - UK	45-50	-	7.5 -9.0	UKTAG (2008a)
Japan	-	2	10	MoEJ (1997)
South Africa	40	-	-	DWAF (1996a,b)
UNECE	< 30 or > 150	3	-	UNECE (1994)
US	-	3 – 5.5	-	US EPA (1986)

Proposed benchmarks for the oxygen regime (Category 1 and 4)

In conclusion, there appears a general worldwide consensus that DO saturation levels between 80% and 120% most likely indicate that the natural oxygen regime is not or only slightly disturbed and appropriate to support the life of aquatic organisms. Oxygen concentration may be used as benchmark value, but the benchmark should be calculated by taking into account the temperature, pressure and salinity of the system. The proposed benchmark values are applicable for a first assessment of ecosystem quality. It should be emphasized that the given range may be too wide for specific organisms or early life stages of organisms in specific aquatic ecosystems. On the other hand, the natural variations of the DO% may be outside the proposed range in specific (parts of) ecosystems with high integrity.

The lowest reported values for the DO% in different guidelines are in the order of magnitude of 30% - 40%. These values are minimum daily averages in order to prevent disturbance of the day-to-day life of the organisms in the water bodies. The proposal is to apply 30% as the absolute minimum and 150% as a maximum. This value corresponds with about 3 mg/l and 13.6 mg/l at sea level and 20 ^{0}C. Although the minimum concentration of DO to maintain higher forms of life varies markedly, the lower benchmark value of 3 mg O2/l represents a value where most multicellular organisms cannot survive for very long. For BOD_5 10 mg/l is proposed as the benchmark for water where aquatic life is extremely impaired. In »Table A 3 the proposed benchmark values are summarized.

Table A 3: Proposed benchmark values for the oxygen regime
[1] Dissolved oxygen concentrations vary depending on temperature, pressure and salinity; benchmark for freshwater at sea level (760 mm Hg) and 20^{0}C based on the DO%
[2] Daily averages

	High Integrity (Cat 1)	Extreme Impairment (Cat IV)
DO Saturation (%)	80 - 120	< 30 or > 150
DO Concentration (mg/l)	7.3 - 10.9 [1]	<3 or > 13.6 [1,2]
(optional) BOD_5 (mg/l)		>10

Nutrients and Chlorophyll

Phosphorus and nitrogen are nutrients essential to aquatic life. However, an excess of nitrogen and phosphorus, thus a kind of chemical pollution can stimulate the growth of algae and higher plants which can dominate and change the structure of aquatic ecosystems. An increase of the nutrient concentrations by anthropogenic influences resulting in “eutrophication” can lead to a series of direct and indirect negative effects such as the stimulated growth of open-water phytoplankton and attached filamentous and “epiphytic” plants, shifts towards dominance of cyanobacteria that include toxic forms, or abundant growth of submerged aquatic plants (macrophytes). In this way the originally chemical pollution can trigger biological pollution of the recipient water. High abundance of photosynthesizing communities can lead to high, even daily, fluctuations in oxygen from super saturation to oxygen depletion, and associated fluctuations e.g. of pH, reduced transparency of standing waters, clogging of water ways, and a large range of indirect shifts in prevalence of ecosystem components affecting both plants and animals. Chl-a is a color pigment that is used by plants, algae and cyanobacteria to utilize sunlight for biomass production. The Chl--a concentration is often used as a general indicator for the biomass of algae and cyanobacteria and as an indicator for the trophic state of a water body.

A wide range of nutrient and Chl-a (and, less commonly, other chlorophyll pigments) concentrations are reported in natural aquatic systems. Natural sources of phosphorus and nitrogen from the leaching of rocks and decomposition of organic matter result in very low “background” concentrations of these nutrients, which are rapidly taken up and incorporated into ecosystem biomass. However, water bodies that drain geology containing phosphorus-rich rocks represent an exception of this rule of thumb. With increased human use of fertilizer, land disturbance, urbanized concentration of humans, and nutrient-rich gaseous, aerosol and particulate emissions to the atmosphere, nutrient enrichment of water bodies has become an increasing problem. Common inputs include those from sewage, industrial effluents, urban runoff, drainage from agriculture land, and atmospheric precipitation or direct deposition polluted with nutrients from vehicles, combustion, livestock and bacterial-mediated denitrification associated with a range of land-uses and industrial processes.

The Australian and New Zealand guidelines (ANZECC/ARMCANZ, 2000b), the Canadian WQGs (CCME, 2004), the European Commission (EC, 2009b) and the South African Guidelines (DWAF, 1996a,b) present valuable general background information about nutrients in water. US EPA published detailed approaches for estimating and interpreting stressor-response relationships for developing numeric criteria to address nitrogen/phosphorus pollution (US EPA, 2010b). More detailed background information about algae, phytoplankton and Chl-a is stated in Fitch and Kemker (2014).

Existing quality criteria and standards for Nutrients and Chlorophyll a (high integrity)

The joint Australian and New Zealand Guidelines provide trigger values for total nitrogen, total phosphorus and Chl-a for upland and lowland rivers, lakes and reservoirs , wetlands, estuaries and marine environments for five geographical regions. The values are derived from unmodified or slightly-modified ecosystems. The trigger values for total phosphorus are between 10 and 25 µg/l for upland rivers in the five geographic regions and between 10 and 100 µg/l for lowland rivers. The range of trigger values for lakes is somewhat smaller: between 10 and 25 µg/l. The trigger values for dissolved phosphorus are about a factor two lower. The total nitrogen ranges in the five geographic regions for upland rivers between 150 and 450 µg/l, for lowland rivers between 200 and 1200 µg/l. The trigger value for lakes is at 350 µg/l in 4 regions of the joint Australian New Zealand Guidelines and at 1000 µg/l in the remaining fifth geographical region. Chl-a concentrations for lakes and lowland rivers are between 3 and 5 µg/l. Chl-a concentrations for upland rivers are not applicable, as open-water populations of algae or cyanobacteria cannot accumulate. Trigger values for lowland rivers and lakes vary between 3 and 5 µg/l.

The Canadian WQGs provide numerical values for total phosphorus and the nitrate-ion concentration. Phosphorus trigger values (CCME, 2004) are based on categories according to the trophic status of high quality reference sites. The internationally accepted OECD trophic state values (Vollenweider and Kerekes, 1982) are also accepted as the Canadian trigger ranges. The trigger range for oligotrophic systems is 4-10 µg/l, for meso-trophic and meso-eutrophic systems 10-35 µg/l and for eutrophic systems 35-100 µg/l. The guideline for the nitrate-ion concentration has been derived with mostly no-effect and some low-effect data during long-term exposure (CCME, 2012). The value of 3000 µg NO_3^--N/l is only for protection from direct toxic effects and does not consider indirect effects due to eutrophication.

In China surface water standards for Total Phosphorus (TP) and Total Nitrogen (TN) are established. In »Table A 4 the standards for the highest quality class (grade I) are presented.

The EU WFD (EC, 2000) presents in Annex V only narrative criteria for the nutrients and phytoplankton blooms. High status requires that nutrient concentrations remain within the range normally associated with minimally disturbed conditions and planktonic blooms occur at a frequency and intensity which is consistent with the type specific physico-chemical conditions. The EU-Member States are responsible for the derivation of standards for the specific ecosystems. The European Commission published an extensive guideline on eutrophication assessment (EC, 2009). The use of empirical data, the application of regression analyses and statistical methods and the estimation of background levels by hindcasting are among the recommended approaches. The UK standards are derived only for phosphorus. It is argued that the general understanding of nitrogen is insufficient at present for it to be used as a basis for setting standards or conditions. The class boundary for high status of phosphorus is 20-30 µg/l for rivers with low alkalinity and 50µg/l for rivers with high alkalinity. The range for lakes is between 5 and 25 µg/l for most types of lakes, but (very) shallow lakes with high alkalinity show values up to 55 µg/l.

The Japanese EQSs for conservation of natural environment (class I) in lakes and reservoirs are TP ≤5 µg/l and TN ≤ 100 µg/l.

In the South Africa WQG average summer concentrations for TP and TN are given for four trophic classes. The values for the oligotrophic status are presented in »Table A 4. The TWQR for inorganic phosphorus and nitrogen concentrations is defined in relationship to un-impacted conditions: concentrations should not deviate by more than 15 % from un-impacted conditions. Moreover, the trophic status should not increase above the present level and the amplitude and frequency of natural cycles not be changed.

The UNECE states WQS for TP, TN and Chl-a for five quality classes. The highest level (Class I) corresponds to clear, oligotrophic water with, at most, a very slight, occasional anthropogenic pollution with organic matter. Total phosphorus should be < 15µg/l for flowing waters and <10 µg/l for lakes and TN below 300 µg/l both for flowing and standing waters. Related indicative Chl-a concentrations are < 4 µg/l for flowing waters and 2.5 µg/l for lakes.

The US EPA has released several peer-reviewed ecoregional nutrient criteria documents for both, rivers and streams and also for lakes and reservoirs (US EPA 2000a, 2000b) including criteria for causative (phosphorus and nitrogen) and responsive (Chl-a and turbidity) variables. The EPA's nutrient criteria represent conditions of surface waters that have minimal impacts caused by human activities. The summary tables show a rather wide range of criteria for twelve ecoregions. The criteria for lakes and reservoirs in the ecoregions are for TP between 8 and 37.5 µg/l, for TN 100 - 1270 µg/l and for Chl a 1.9 – 12.35 µg/l. Most of the values are at the lower side of these ranges: in 9 of 12 ecoregions TP is proposed to be at < 20 µg/l, TN at < 560 µg/l and Chl-a at < 3.5 µg/l. The criteria for rivers and streams are for TP between 10 and 76 µg/l, for Total TN 120 – 880 µg/l and for Chl a 0.4 – 3.75 µg/l. For two regions the values largely deviate from this range: for one region is TP 128 µg/l and for another region TN is 2180 µg/l.) Moreover, most of the regions show values at the lower side of the range: TP < 40 µg/l and TN < 700 µg/l. Generally the range in nutrient concentrations in rivers is larger than in lakes because of the settlement of particulate phosphorus associated with larger water residence time of lakes, while the range of Chl-a concentration in rivers is slightly smaller and the criteria are lower than in lakes, again reflecting effects of water residence time. The recommended values are suggested baselines which states and tribes should refine to help identify problem areas. In 2010 a supplementary document was published which provides detailed approaches for estimating and interpreting stressor-response relationships for developing numeric criteria to address nitrogen/phosphorus pollution (US EPA, 2010b). Several statistical models are used to model the relationship between nutrient concentrations and Chl-a concentrations. See »Table A 4 for an overview of nutrient and Chl-a criteria for high ecosystem integrity in a number of countries and entities.

Table A 4: Overview of some Water Quality Criteria (WQC) for Nutrients and Chlorophyll a (Chl-a) in for high ecosystem integrity in freshwaters.
[1] For protection from direct toxic effects; the guideline do not consider indirect effects due to eutrophication

	Total P in µg/l	Dissolved P in µg/l	Total N in µg/l	Nitrate-N in µg/l	Chl-*a* in µg/l	Reference
Australia/ New Zealand - Lakes - Rivers and streams	 10 - 25 10 - 100	 5 - 10 4 - 40	 350 - 1000 150 - 1200	 10 - 100 15 - 444	 3 - 5 3 - 5	ANZECC/ARMCANZ (2000a)
Canada	4 - 10	-	-	(3,000)[1]		CCME (2004) CCME (2012)
China	20	-	200	-	-	China (2002)
EU - UK - Lakes - Rivers	 5 – 55 -	 - 20 - 50	 - -	 - -	 - -	 UKTAG (2008b) UKTAG (2008a)
Japan	5		100			MoEJ (1997)
South Africa	-	5	-	500	-	DWAF (1996a,b)
UNECE - Lakes - Flowing waters	 10 15	 - -	 300 300		 2.5 4	UNECE (1994)
US - Lakes and reservoirs - Rivers and streams	 8 - 37.5 10 - 76	 - -	 100 - 1270 120 - 880	 - -	 10.9 – 12.35 0.4 – 3.75	US EPA (2002)

Existing quality criteria and standards for Nutrients and Chlorophyll a (Extreme Impairment)
The Canadian WQGs present TP trigger values based on the categories published by the

OECD (Volleweider and Kerekes, 1982). Extreme eutrophic systems, so-hyper-eutrophic systems have values of TP >100 µg/l. The guideline for nitrate-N concentration is estimated at 124,000 µg/l. The nitrate value is derived from severe effects data (such as lethality), but does not consider indirect effects due to eutrophication. Below the value most species are only protected against lethality during short-term exposure.
For China the standards for the lowest quality class (Grade V) are presented in »Table A 5.

Based on the EU WFD, the UK standards provide phosphorus concentration in poor systems. The values for soluble reactive phosphorus are > 500 µg/l in low alkalinity rivers and > 1,000 µg/l for high alkalinity rivers.
In Japan the lowest class of uses corresponds with concentrations of TP ≤ 100 µg/l and TN ≤ 1,000 µg/l. Above these values the day to day-lives of the population may be limited or disrupted.

The boundaries for the lowest class of surface water quality published by UNECE (1994) are TP > 190 µg/l for flowing waters and > 125 µg/l for lakes, TN > 2,500 µg/l and Chl-a > 165 µg/l for flowing waters and > 110 µg/l for lakes and reservoirs. These concentrations indicate extensively polluted, hypertrophic water.
In the South-Africa WQGs levels of TP > 250 µg/l and TN > 10,000 µg/l as average summer concentrations refer to hypertrophic conditions with usually nuisance growth of aquatic plants and blooms of cyanobacteria, often including species that are toxic to man, livestock and wildlife.

Table A 5: Overview of some Water Quality Criteria (WQC) for total phosphorus, dissolved phosphorus, total nitrogen, nitrate-N and Chl-a for extreme impairment of freshwater ecosystems.
[1] For protection from direct toxic effects; the guideline do not consider indirect effects due to eutrophication

	Total P in µg/l	Dissolved P in µg/l	Total N in µg/l	Nitrate-N in µg/l	Chl-*a* in µg/l	Reference
Australia/ New Zealand	-	-	-	-	-	ANZECC/ARMCANZ (2000a)
Canada	100	-	-	(124,000)[1]		CCME (2004) CCME (2012)
China	400	-	-	2000	-	China (2002)
EU-UK - Lakes - Rivers	 - -	 - 500 - 1000	 - -	 - -	 - -	UKTAG (2008a)
Japan	100		1000			MoEJ (1997)
South-Africa		250		10000		DWAF (1996a,b)
UNECE - Lakes - Flowing waters	 125 190	 -	 2500 2500	 - -	 110 165	UNECE (1994)
US	-	-	-	-	-	US EPA (2016a)

Proposed benchmarks for Nutrients and Chlorophyll a (Category 1 and 4)
It may be concluded from the guidelines considered that the setting of a general benchmark for nutrient and Chl-a for high integrity of ecosystems is difficult to establish. Although there is a huge amount of data available concerning the relationship between nutrient concentrations and phytoplankton growth (expressed as Chl-a), natural concentrations of nutrients as well as the relationship between nutrient and Chl-a under un-impacted conditions vary. Besides differences in the natural trophic state owing to natural sources of nutrients and geographic factors, the relationship between nutrients and Chl-a are influenced by a range of factors, including alkalinity, transparency, flow conditions and water depth. Most of the guidelines strongly emphasize the importance of taking into account natural site-specific conditions for an assessment of the influence of nutrients and the setting of criteria. Nevertheless, a number of guidelines provide numerical criteria for nutrients and Chl--a, with differences often made between rivers and lakes. Based on existing guidelines the proposed benchmark values for Category 1 are for TP < 10 µg/l for lakes and reservoirs TP < 20 µg/l for rivers and streams, for TN < 500 µg/l and < 700 µg/l respectively and for Chl a < 3 and < 5 µg/l. These concentrations refer to a state where nutrients may be expected to be low and have minimal impact; although in some situations even these seemingly low values may already signify some ecological impact.

Some guidelines present values for dissolved phosphorus and nitrate-N instead of total concentrations. Both indicators might be better indicators for the trophic status and the impact of eutrophication but both are less often used in the guidelines considered. The differences between total phosphorus and dissolved phosphorus are mostly relatively small. The difference between total nitrogen and nitrate-N may be rather large. As shown in »Table A 6 the existing guidelines present strongly varying values for nitrate-N. For that reason no proposals are given for dissolved phosphorus and nitrate-N.
In the guidelines considered only a few trigger values are given for systems in extremely impaired quality status. The UNECE values of 1994 are propposed as benchmarks, unless new scientific-based guidelines are available.

The WHO guideline values for drinking-water (WHO, 2011) are for nitrate 50.000 µg NO_3^- /l and for nitrite 3000 µg NO_2^- /l. Both WHO- values are much higher than the proposed benchmark for total nitrogen levels for extreme impairment. In consequence, concerning nitrogen meeting the benchmarks values for the protection of aquatic life implies also the protection of drinking-water sources.

Table A 6: Proposed benchmark values for Nutrients and Chlorophyll a
[1] Background concentrations in undisturbed natural waters may deviate due to natural nutrient sources and geographical conditions.

	High Integrity (Cat 1)[1]	**Extreme impairment (Cat IV)**
Total Phosphorus (µg/l)		
- lakes and reservoirs	**< 10**	**>125**
- rivers and streams	**< 20**	**>190**
Total Nitrogen (µg/l)		
- lakes and reservoirs	**< 500**	**> 2500**
- rivers and streams	**< 700**	**> 2500**
Chlorophyll *a* (µg/l)		
- lakes and reservoirs	**< 3.0**	**> 165**
- rivers and streams	**< 5.0**	**> 125**

pH
The pH-value of water indicates how acid or basic a water body is. The scale is a logarithmic one from 0 (extreme acidic) and 14 (extremely basic), hence each increment of pH represent an order of magnitude in acidity. A pH of 7 is considered as neutral, but pure water has a slightly lower value. Normal values for freshwater bodies vary between 6 and 9. Very low or high pH values, affects cellular function of most aquatic organisms, resulting in reduced growth, reproduction or survival. The pH is also highly relevant because the solubility and toxicity of several heavy metals vary with pH. A change in pH can also alter availability of nutrients for algae growth in the water column and the release of nutrients from the sediment. Values of ambient pH values are affected by geological, hydrological and biological processes. Carbonate material and limestone can lead to higher pH values and can buffer pH changes. If biological decomposition processes dominate pH values can decrease. High photosynthesis rates increase the pH values. In waterbodies with high rates of respiration and decomposition significant diurnal variations in pH occur. In eutrophic systems the pH may fluctuate widely within 6 and 10 over a 24-hour period. Differences in temperature have only very slight effects on pH values. Human induced causes of pH changes include low-pH point-sources, mine-drainage, and acid precipitation as a result of emissions of sulphur dioxides and nitrogen oxides. More indirectly, nutrient enrichment may cause large variations of pH.

The alkalinity of water is a measure for the ability to neutralise acids, the so-called buffer capacity. The alkalinity of water depends on the concentrations of dissolved salts and carbonates in the water. Dissolution of carbonate-rich rocks and soils in the catchment area is the main source for high alkalinity. Alkalinity is important for fish and aquatic life because it protects or buffers against rapid pH changes. If water has a high alkalinity, pH changes due to anthropogenic influences are limited. Low alkalinity of a water body may cause clear changes in the pH by acid rain and acid discharges.

In humic waters the decomposition of organic matters is generally the dominant process. Humic waters are water bodies with high levels of Dissolved Organic Carbon (DOC). Humic acids are produced by decomposition processes and decrease pH.

Spread across the world there are a number of lakes with unusual pH values (Kemker, 2013b). On the one hand, alkaline lakes, also known as soda lakes, generally have pH values between 9 and 12. These lakes have high contents of minerals, particularly dissolved salts, but can also be important sites for e.g. flamingo breeding. On the other hand, there are naturally highly acid lakes, which have usually been developed near volcanoes. In some of these can even be found fish with special adaptation to cope with acidity as low as 3.5.

More detailed background information about the relevance of pH and alkalinity, about the relation between the two and about the source cause of natural and anthropogenic differences in pH values, as well as diurnal variations in pH values is Kemker (2013b). The Australian and New Zealand guidelines (ANZECC/ARMCANZ, 2000b) and the South African Guidelines (DWAF, 1996a,b) present additional valuable background information about the pH in water.

Existing quality criteria and standards for pH (high integrity)
The joint Australian and New Zealand Guidelines provide trigger values for pH values for five geographical regions across Australia and New Zealand, including upland and lowland rivers and lakes. Nearly all pH values are between 6.0 and 9.0, but in a number of regions the values are within tighter constraints.

The WQG for the protection of live in Canada recommends a pH in the range of 6.5 - 9.0. In China the surface water standard for pH is 6 - 9 for all classes of water. The EU WFD presents in Annex V only narrative criteria for the pH condition. The high status is described as follows: “the pH and acid neutralizing capacity (= alkalinity) do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions”. The EU-members are responsible for the derivation of standards for the specific ecosystems. In the UK, recommended standards for rivers are derived for clear waters and humic waters. For rivers the recommended high quality class standard for pH in clear waters is > 6.6 and for humic waters > 5.1.

In Japan the EQSs for conservation of the natural environment (Class AA) for pH is 6.5 < pH < 8.5 both for rivers and lakes.

In the South African WQGs the TWQR for pH is established as a deviation of the background pH value. Hence, pH values should not be allowed to vary from the range of background values for a specific site or time of the day by < 0.5 of a pH unit or by < 5%. WQSs for the pH for five classes are given by UNECE. At the highest level (Class I) the values for pH are 6.6 - 9.0. Class I is described as “waters with a very good buffering capacity”.

The US EPA water criteria gives a range for pH from 6.5- 9 for long-term exposure. An overview of pH values for high ecosystem integrity in quality guidelines in a number of countries and entities is given »Table A 7.

Table A 7: Overview of some Water Quality Criteria (WQC) for pH for high ecosystem integrity in freshwaters.
[1] Value used for humic waters

	pH	Reference
Australia/ New Zealand	6.0 – 9.0	ANZECC/ARMCANZ (2000a)
Canada	6.5 – 9.0	CCME (1999)
China	6.0 – 9.0	China (2002)
EU - UK	6.6 (5.1[1])	UKTAG (2013)
Japan	6.5 – 8.5	MoEJ (1997)
South Africa	0.5 deviation	DWAF (1996a,b)
UN ECE	6.5 - 9.0	UNECE (1994)
US	6.5 – 9.0	US EPA (1986)

A few guidelines present values for alkalinity, because alkalinity provides insights to the buffer capacity of water. As alkalinity is mainly determined by natural processes and the relation between alkalinity and aquatic life is rather weak in reasonable buffered systems, no benchmark is stated in this document for alkalinity for high integrity ecosystems (Category 1). However, alkalinity can be of great use when a typology of water bodies is developed, i.e. when water bodies are grouped in coherent evaluation and management units based on their structure and function.

Existing quality criteria and standards for pH (extreme impairment)
The WQGs for aquatic life of Australia/New Zealand do not provide values for minimum or maximum pH values, but the guidelines refer to reviews which indicate that there are no acutely lethal effects in the range of 5 – 9 and chronic effects have been reported below 5 mg/l. In other studies it was found that spawning failure and diminished hatching success may occur below pH of 6.

The UK standards for the classification of poor water quality are pH < 4.89 for clear waters and pH < 4.03 for humic waters.

In Japan the minimum quality standard for pH has been established at 6.0 ≤ pH ≤ 8.5 mg/l both for rivers and lakes. Kemker (2013b) presents recommended minimum pH levels for aquatic life. Most species require pH levels higher than 5 or more except some fish species and species of amphibians which often can tolerate lower values. The boundary for the lowest class of surface water quality published by UNECE (1994) is a pH of < 5.3. Acidity is generally considered to be toxic to most multicellular organisms below this value.

An overview of pH values for extreme impairment in quality guidelines in a number of countries and entities is given »Table A 8.

Table A 8: Overview of some Water Quality Criteria (WQC) for pH for extreme impairment of freshwater ecosystems.
[1] Humic waters

	pH	Reference
Australia/ New Zealand	< 5.0	ANZECC/ARMCANZ (2000a)
Canada	-	CCME (1999)
China	< 6.0 or > 9.0	China (2002)
EU - UK	< 4.89 (<4.03[1])	UKTAG (2013)
Japan	< 6.0 or < 8.5	MoEJ (1997)
South Africa	-	DWAF (1996a,b)
UN ECE	< 5.3	UNECE (1994)
US	-	US EPA (1986)

Proposed benchmarks for pH (Category 1 and 4)
Summarizing the criteria and standards in the selected guidelines the range of pH values between 6.5 and 9.0 are mostly accepted as reasonable values for ecosystems with high integrity. However, natural geochemical and biological processes may cause pH values outside this range. E.g. so-called alkaline lakes have rather high pH levels and humic waters and volcanoes lakes and rivers may show pH values below 6.5.

As a provisional conclusion, pH values lower than about 5 may be harmful for certain kinds of aquatic live. However, the information about minimum and maximum values indicating very poor aquatic systems is rather scarce. A limited number of species can survive under extreme pH conditions.

Table A 9: Proposed benchmark values for pH
[1] Because of the logarithmic scale for pH the annual average cannot be calculated directly by simple averaging. It can be reported as range or mean.
[2] Humic waters and acid lakes have naturally lower pH values
[3] Alkaline lakes have naturally higher pH values

	High Integrity (Cat 1)	**Extreme impairment (Cat4)**
pH[1]	**6.5[2] – 9.0[3]**	**< 5**

Temperature
The temperature of surface waters influences variety of aquatic life. Some organisms prefer warm waters and other organisms can only survive in cold waters. The temperature and the fluctuations in temperature of the water influence the growth, metabolisms, reproduction success, and mobility as well as migration patterns. The thermal tolerance of species can differ markedly. The rate of photosynthesis and microbiological processes will generally increase with increasing temperature, but temperature can also influence aquatic life indirectly, as the temperature can alter physical and chemical properties of the water, e.g. the DO concentration, the solubility and bioavailability of toxic compounds like heavy metals and the relative amount of unionized ammonia.

The temperature of surface waters can vary largely, from frozen to above 35 °C. The main determining factor is solar radiation, but also differences in temperature between the water and air, the temperature of the water sources and turbidity may influence temperature. Stratification of waters leads to vertical differences in temperature. Thermal stratification may occur seasonally in lakes.

Human causes of temperature changes include discharges of heated industrial effluents and power stations, heated discharges of sewage waters, heated return flows of irrigation waters and removal of riparian vegetation cover. Shifts in climate, already showing impacts in distribution of aquatic life, are likely to be more prevalent over the next few decades.

More detailed background information about the relevance of temperature for aquatic life and natural and anthropogenic variations in temperature is stated in Kemker (2013a). The Australian and New Zealand guidelines (ANZECC/ARMCANZ, 2000b) and the South African Guidelines (DWAF, 1996a,b) present valuable background information about the temperature of water.

Existing quality criteria and standards for Temperature (high integrity)
The joint Australian and New Zealand Guidelines provide no general trigger values for temperature, but advice for Category 1 ecosystems (high conservation/ecological value) that there should be no change from ambient conditions, unless it can be demonstrated that such a change will not compromise biological diversity. Low risk trigger values can be derived on the basis of biological and ecological effects, reference system data, predictive modelling and expert judgement. The guideline indicates that there is little information on the thermal tolerance of Australian and New Zealands´ aquatic organisms.

The Canadian guidelines do not provide numerical values, but e.g. in the province British Columbia WQG values for temperature are estimated, both as mean weekly maximum temperature and as temperature deviation from optimum temperature ranges of specific life history stages of salmonids and other cold water species (Gov BC, 2001). The allowed changes are +1 or -1 ^{0}C from ambient background values or from the optimum temperature range of specific life history stages of salmonids and other cold water species.

In China a surface water standard for temperature is equal for all classes of water: man-made fluctuations should be limited: maximum weekly average temperature rise is at ≤ +1 ^{0}C and the maximum weekly average temperature drop is at ≤ 2 ^{0}C.

Table A 10: Overview of some Water Quality Criteria (WQC) for temperature for high ecosystem integrity in freshwaters.
[1] Values from the Government of British Columbia, a province of Canada

	Temperature	Reference
Australia/ New Zealand	No change from reference conditions	ANZECC/ARMCANZ (2000a)
Canada	+1 or -1 ^{0}C from ambient natural background[1]	CCME (1999)
China	Man-made fluctuations maximum weekly average rise ≤ 1 and weekly average drop ≤ 2 ^{0}C	China (2002)
EU - UK	< 25 ^{0}C for warm waters and < 20 ^{0}C for cold waters	UKTAG (2008b)
Japan	-	
South-Africa	Deviation of +2 ^{0}C and -2°C or 10% from ambient natural background on an average daily temperatures	DWAF (1996a,b)
UN ECE	-	UNECE (1994)
US	-	USEPA (2016a)

The EU WFD presents in Annex V only a narrative criterion for the temperature: “the temperature does not show signs of human disturbance and remain within the range normally associated with natural variation". The EU-Member States are responsible for the derivation of standards for the specific ecosystems. In the UK, recommended standards for rivers are specified for cold waters and warm waters. The proposed boundaries for high quality are 20^{0}C for cold water and 25^{0}C for warm water. At this level expected type-specific fish species are present and their abundance is consistent with undisturbed conditions in the rivers in the UK.

In the South Africa WQGs the TWQR for water temperature for all aquatic ecosystems should not be allowed to vary from the background average daily water temperature considered to be normal, i.e. typical for the specific site and time of day, by > 2^{0}C, or by > 10 %, whichever estimate is the more conservative.

The US EPA water criteria do not provide numerical values for temperature. But the US WQC 1986 (US EPA, 1986) provides data from studies documenting the thermal tolerance of various fish species, e.g. data to identify temperatures associated with lethal exposures for species as well as temperate ranges associated with growth and spawning periods. The data are however presented as a tool to derive criteria for the evaluation thermal discharges and not as a tool for the assessment of ambient water temperature (»Table A 10).

Existing quality criteria and standards for Temperature (extreme impairment)
For highly disturbed ecosystems the WQGs for aquatic life of Australia/New Zealand recommend site-specific studies and professional judgement to derive trigger values. Especially if the values are outside the range 20th or 80th percentile values of appropriate reference systems further ecosystem-specific investigation is recommended (»Table A 11).

In China a surface water standard for temperature is equal for all classes of water: man-made fluctuations should be limited: maximum weekly average temperature rise is at ≤ 1 ^{0}C and the maximum weekly average temperature drop is at ≤ 2 ^{0}C.
In the UK, recommended standards for rivers based are specified for cold waters and warm waters. The proposed boundaries for poor quality are 30^{0}C for cold water and 32^{0}C for warm water. Poor quality means that the communities deviate substantially from those normally associated with the waterbody and/or that key species may be absent.
The US EPA (1986) adopted a formula to determine the upper thermal limits for heated effluent discharges based on thermal optima for a number of fish species.

Table A 11 Overview of some Water Quality Criteria (WQC) for temperature for extreme impairment of freshwater ecosystems.

	Temperature	Reference
Australia/ New Zealand	Outside the range 20th or 80th percentile of reference conditions	ANZECC/ARMCANZ (2000a)
Canada	-	CCME (2016)
China	Man-made fluctuations maximum weekly average rise ≤ 1 and weekly average drop ≤ 2 ^{0}C	China (2002)
EU - UK	*< 32 ^{0}C for warm waters and < 30 ^{0}C for cold waters*	UKTAG (2008b)
Japan	-	MoEJ (1997)
South-Africa	-	DWAF (1996a,b)
UN ECE	-	UNECE (1994)
US	-	USEPA (2016a)

Proposed benchmarks for Temperature (Category 1 and 4)
Summarizing, general numerical criteria for temperature in aquatic ecosystems are not presented in the guidelines considered. Two ways are used for the assessment of temperature changes. Firstly, deviations from background values or reference sites may be used. For the high integrity benchmark the proposed criterion is that there are no deviations. Secondly, a benchmark may be derived from the optimum temperature range for specific life stages of relevant species (»Table A 12).

The guidelines considered in this annex provide only very limited information concerning numerical temperature criteria for extreme impairment of ecosystems. Benchmark values might be derived based on large deviations from background values or clear deviations from the thermal tolerance of characteristic species in an aquatic ecosystem. Further information in »Box A 1.

Table A 12: Proposed benchmarks for Temperature

	High Integrity (Cat 1)	Extreme impairment (Cat 4)
Temperature	No deviation from background value or reference systems or optimum temperature ranges of relevant species	Large deviations from background value or the thermal tolerance range for characteristic species

Temperature is a fundamental water quality parameter, which controls to a great degree the physical, chemical, biological, and ecological processes and functions in streams and rivers. Water temperature is the result of complex interactions of climatic, hydrological and landscape processes as well as channel properties. Changes in river temperature occur naturally or result from human-induced alterations such as climate change, deforestation, flow alteration, release of effluents, and damming. Alterations of the thermal regime may have severe consequences for biodiversity, ecosystem processes, and the related services such as drinking water production, fisheries, and recreation.

Conventional standards of water quality are based on simple thresholds, while regime-based standards cover the distribution of quality components over space and time. For example, the temperature regime of rivers can be described in terms of magnitude, variability, frequency, duration, and timing of thermal events (e.g. Arismendi et al., 2013, Dallas and Rivers-Moore, 2012). Indeed, temperature should be integrated into water quality monitoring and assessment programs, and there is a clear need to identify the river-specific thermal components that need to be considered.

Box A 1: Thermal Regime

Ammonia

Ammonia is one of the several forms of nitrogen that exist in aquatic environments. Ammonia occurs in two forms in the aquatic environment: the un-ionized form (NH_3) and the ionized form (NH_4^+). The ionized form is also referred to as ammonium. The Total Ammonia Nitrogen (TAN) is the sum of both forms. The ionized ammonia and the un-ionized forms are interrelated through the chemical equilibrium $NH_4^+OH^-$ <-> NH_3H_2O <-> $NH_3 + H_2O$.

The relative concentrations of NH_4^+ and NH_3 are basically dependent on the pH and temperature. With increasing values of pH and temperature, the concentration of NH_3 increases and the concentration of NH_4^+ decreases. Ammonia in aquatic systems is converted into less toxic nitrate by nitrifying bacteria and the use of oxygen.

Unlike other forms of nitrogen in aquatic systems, ammonia causes direct effects on aquatic life. Especially the un-ionized form NH_3 is very toxic to aquatic organisms, whereas in the ionized form it is nontoxic. There are substantial data available on the toxicity to acute, lethal and sub-lethal effects of ammonia on fish, and effects on invertebrates and benthic organisms are also described. As NH_3 causes harmful effects, increase of pH or temperature has negative effects on aquatic life due to an increase of the concentration of NH_3. Beside the toxic effect, discharges of ammonia also contribute to eutrophication problems. Further information is given in the Section on nutrients and Chl-a of this annex. Large discharges of ammonia may also cause oxygen depletion.

Natural sources of ammonia include the decomposition of organic material, animal waste and discharges of ammonia by biota. The most important point sources include emissions and effluents from industrial plants, sewage treatments plants and accidental ammonia

spills. The non-point sources include agricultural, municipal and atmospheric releases. Major agricultural sources include run-off in areas with intensive farming, accidental release or spills of fertilizer and the decomposition of livestock waste.
including ammonia, has been done by Camargo and Alonso (2006).

Existing quality criteria and standards for Ammonia (high integrity)
The joint Australian and New Zealand Guidelines state a trigger value ammonia of 320 µg/l for total ammonia-N calculated at a pH level of 8.0 using the statistical distribution method with 99% level of protection of species. A table indicates how the guideline figure changes at different values of pH and temperature.

The long-term concentrations for the protection of aquatic life based on the Canadian Environmental Quality Guidelines are shown in ».Table A 13 These long-term exposure guidelines identify benchmarks that are intended to protect all forms of aquatic life (all species, all life stages) for indefinite exposure periods.

The Chinese guideline include a standard of 150 µg/l for NH_4^+ for grade I waters. However, the corresponding pH-value is not specified. Moreover, the way of how it was derived was not published in English. Therefore, it is rather difficult to compare the Chinese standard value with criteria and standards from other countries and regions using rather the (un- ionized) ammonia in their standards.

In the UK standards for total ammonia have been developed on the basis of toxicity of un-ionized ammonia. For upland and low alkalinity freshwaters the proposed EQS is 200 µg/l and for lowland and high alkalinity fresh waters 300 µg/l (UKTAG, 2008C). These are the 90-percentage values, which means that the values exceeded for 10 per cent of the time. In the updated recommendations (UKTAG, 2013) these values have been confirmed. Moreover, for un-ionized ammonia a 90-percentile EQS of 40 µg/l has been proposed.
In the South-Africa WQGs the TWQR for un-ionized ammonia is 7.0 µg/l. The TWQR is a management objective to specify the desired or ideal concentration range. The TWQR is based on a CEV of 15 µg/l. The guideline includes a table for the calculation of the contribution of un-ionized NH_3 to measure total ammonia as a function of pH and temperature.

The US EPA has reviewed the ammonia criteria in 2013 (US EPA 2013a). US EPA recommends a chronic criterion magnitude of 1900 µg/l TAN at pH 7 and 20 ^{0}C for a 30-day average duration, not to be exceeded more than once every three years on average. In addition, the highest 4-day within a 30-day period should not exceed 2.5 times the chronic magnitude more than once in three years on average.

Table A 13: Overview of some Water Quality Criteria (WQC) for ammonia for high ecosystem integrity in freshwaters.
[1] At pH 8.0
[2] Calculated value based on 320 µg/l TAN at pH 8.0 and 20 ^{0}C
[3] At pH 7.0 and 20^{0}C
[4] Calculated value based on 1900 µg/l TAN at pH 7.0 and 20 ^{0}C
5 NH_4^+, pH unknown; not published in English

	Total Ammonia Nitrogen (TAN) in µg/l	Un-ionized Ammonia in µg/l NH_3	Reference
Australia/ New Zealand	320[1]	(14.8)[2]	ANZECC/ARMCANZ (2000a)
Canada	-	19	CCME (2010)
China	(150)[5]	-	China (2002)
EU-UK	200 - 300	40	UKTAG (2008a,b), UKTAG(2013)
Japan	-	-	MoEJ (1997)
South Africa	-	7	DWAF (1996a,b)
UN ECE	-	-	UNECE (1994)
US	1900[3]	(9.1)[4]	US EPA (2013a)

Existing quality criteria and standards for ammonia (extreme impairment)

The joint Australian and New Zealand Guidelines provide trigger values for total ammonia at a 80% level of protection of species at 2300 µg/l of ammonia (»Table A 14). This level is associated with very highly disturbed systems. Even a 90% level of protection is referred to as “highly disturbed”.

The Chinese guideline includes a standard of 2000 µg/l for NH_4^+ for the lowest class (grade V). However, the corresponding pH-value is not specified. Moreover, the way of deriving was not published in English. So, it is rather difficult to compare the Chinese standard with criteria and standards for un-ionized ammonia from other countries and entities.

In the South-African WQGs an AEV is defined as a concentration above which there is expected to be significant probability of acute toxic effects to up to 5% of the species in the aquatic community.

Table A 14: Overview of some Water Quality Criteria (WQC) for ammonia for extreme impairment of freshwater ecosystems.

1 At pH 8.0

2 Calculated value based on 2300 µg/l TAN at pH 8.0 and 20 °C

3 At pH 7.0 and 20°C

4 Calculated value based on 1900 µg/l TAN at pH 7.0 and 20 °C

5 NH_4^+, pH unknown; not published in English

	Total Ammonia Nitrogen (TAN) in µg/l	Un-ionized Ammonia in µg/l NH_3	Reference
Australia/ New Zealand	2300[1]	(106.3)[2]	ANZECC/ARMCANZ (2000a)
Canada		-	CCME (2010)
China	(2000)[5]	-	China (2002)
EU-UK	-	-	UKTAG (2013)
Japan	-	-	MoEJ (1997)
South Africa	-	100	DWAF (1996a,b)
UNECE	-	-	UNECE (1994)
US	17,000[3]	(81.7)[4]	US EPA (2013a)

Proposed benchmarks for un-ionized ammonia (Category 1 and 4)

As un-ionized ammonia (NH_3) is the toxic compound of the total ammonia concentration it is proposed to set a benchmark for the un-ionized form. Moreover, the toxic effect of total ammonia depends highly on pH and to a lesser extent on temperature, because the relative amount of NH_3 depends on pH and temperature. E.g. at pH 8.5 the proportion of un-ionized ammonia is approximately 10 times higher than at pH 7.5.

The proposed value for un-ionized ammonia is 15 µg/l, i.e. a value close to the median values of the criteria and standards in the guidelines considered in this annex (see »Table A 15). The percent of un-ionized ammonia of measured concentrations of total ammonia at different pH and temperature situations is stated in Emerson et al. (1975). Information how to calculate NH_3/ NH_4^+ concentrations as function of pH is also included in a number of guidelines e.g. ANZECC/ARMCANZ (2000b), CCME (2010) and DWAF (1996a,b). At a pH of 7.5 and 20°C 15 µg/l NH_3 corresponds to nearly 1000 µg/l total ammonia-N.

Based on »Table A 15 a benchmark for un-ionized ammonia of 100 µg/l NH_3 is proposed for extreme impairment of freshwater ecosystems. This value corresponds with 6641 µg/l total ammonia-N.

Table A 15: Proposed benchmarks for Un-ionized Ammonia

	High Integrity (Cat 1)	Extreme impairment (Cat 4)
Un-ionized Ammonia in µg/l NH_3	15	100

Heavy metals

The term “heavy metals” refers to a group metals which may cause water pollution and adverse biological effects. The metals most frequently mentioned in water pollution studies and in WQGs are aluminum (Al), arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni) and zinc (Zn). Other metals like iron, selenium, silver and manganese are less frequently reported as environmental problems. Metals are different from organic pollutants because they are not created or destroyed by biological or chemical processes. Some metals are essential nutrients important for biological growth, survival and reproduction. Essential metals are Cr, Cu, Ni and Zn, whereas As, Cd, Pb and Hg are non-essential metals (EC, 2011b). Excess amounts of these metals are potentially toxic. An extensive number of studies on the toxic effect on aquatic species has been published and used to derive WQC for heavy metals. The bioavailability, and therefore, the toxicity of heavy metals in aquatic systems are related to the dissolved or sediment-bound fraction of the heavy metals. The dissolved fraction depends on water temperature, pH, alkalinity, salinity and the amount and character of suspended matter. Some forms of metals are more toxic, e.g. Cr VI is more toxic than Cr III and the toxicity of Hg is mainly determined by the prevalence of methyl-mercury. Some metals accumulate in fatty tissues of organisms (bio concentration) and may increase as they are transferred through predation up the food chain (bio magnification). Some heavy metals are bound to suspended matter and occur in sediments and may cause adverse effects for organisms living in or near the riverbed and lake bottom.

Heavy metals are elements which are present in geological structures and can therefore enter water resources by natural processes. Natural background concentration in aquatic systems differs owing to the natural composition of the catchment area and the intensity of leaching processes by rain and other water flows.

The main human sources of heavy metal pollution are discharges by mining activities and by others industrial activities such as metal processing industries and manufacturing of leather, paints, dyes, paper, and ceramics. Non-point sources also contribute to heavy metal pollution: e.g. run-off of heavy metal containing pesticides and fertilizers, corrosion of metallic constructions and pipes, contaminated precipitation caused by e.g. explosives, combustion processes and vehicle emissions.

A brief overview of the impacts of metals on aquatic ecosystems and human health is published by Solomon (2008). A more detailed analysis of the toxic effects is published by Wright and Welbourn (2002). The Australian and New Zealand guidelines (ANZECC/ARMCANZ, 2000b), Canadian Guidelines (2016) and the South African Guidelines (DWAF, 1996a,b) provide valuable background information about each heavy metal in water and the way of deriving of the guideline values.

Existing quality criteria and standards Heavy Metals (high integrity)
The joint Australian and New Zealand Guidelines provide general trigger values for heavy metals. The trigger values are the concentrations below which there is a low risk that adverse biological effects will occur. The presented values in »Table A 16 are the 99% level of protection of species. The trigger values for Cd, Cu, Pb, Ni and Zn metals depend on hardness of the water. The preferred approach of the guideline is to first establish the background concentrations. The guidelines provide information on typical background concentrations (ANZECC/ARMCANZ, 2000b) based on a publication by Hickey and Pyle (2001). Most of the natural background concentrations are reasonably lower than the trigger values, but in some waters the background value may exceed the stated guideline trigger value due to mineralization from the catchment substrate e.g. in volcanic areas. From the Canadian Environmental Quality Guidelines, the long-term concentrations for the protection of aquatic life are shown in »Table A 16. These long-term exposure guidelines identify benchmarks that are intended to protect all forms of aquatic life (all species, all life stages) for indefinite exposure periods. The Cu, PB and Ni values depend on water hardness.

In China the surface water standards for most of the heavy metals are established for the five quality classes. The concentrations stated in »Table A 16. represent the standards for the highest class (grade I).

In accordance with the EU WFD EQSs are established for priority substances and certain other pollutants on community level e.g. for the metals cadmium, lead, mercury and nickel. The methods for deriving EQSs are described in the guidance document No 27 (EC, 2011b). For mercury and its compounds the standard is not established for the concentration in the water, but in organisms: an EQS of 20 µg/kg in prey tissue (wet weight). The objective of the EQS is to set standards for a good surface water chemical status. The values should not be exceeded based on an annual average. EQSs for other metals should be derived by the Member States. In the UK EQSs are derived for, among others, arsenic, copper, chromium and zinc (UKTAG, 2008c). These values are called Predicted No Effect Concentrations values : a concentration of a pollutant below which no harmful effects on aquatic organisms would be expected. Member States may set EQS for sediment and/or biota at a national level according to the guidance document (EC, 2011b) and apply those EQS instead of EQS for water.

The Japan EQSs only provide a standard for zinc.

The WQS for the nine heavy metals for five classes are given by UNECE. Class I is described as waters without anthropogenic pollution with inorganic matter. Therefore, the values may be interpreted as general natural background values.

In the South Africa WQGs TWQR for metals are established. The TWQR is a management objective to specify the desired or ideal concentration range. These presented TWQR are based on the CEV. The TWQRs are mostly a factor 2 or more lower than the CEV. The TWQR for Cd, Cu and Pb depend on water hardness.

The US EPA presents WQC for acute and chronic exposure of aquatic life. In »Table A 16 the chronic values are presented. These values are estimates of the highest concentration to which an aquatic community can be exposed indefinitely without resulting in an unacceptable effect and are intended to be protective of the vast majority of the aquatic communities in the US. The US-values are expressed as dissolved concentrations, because the bioavailability and therefore the toxicity of a particulate metal is substantially lower than that of the dissolved metal. Conversion factors for dissolved metals are given in Appendix A of the Criteria Tables (US EPA, 2016a). Criteria for Cd, Cr, Pb, Ni and Zn depend on hardness of the water. Parameters for calculating freshwater dissolved metals criteria that are hardness-dependent are given in Appendix B of the Criteria Tables (US EPA, 2016). The criteria are derived based on the guidelines for deriving numerical national WQC for the protection of aquatic organisms (US EPA, 1985)

An overview of heavy metal criteria for protecting aquatic life in the nine WQGs considered are presented in »Table A 16. All values are related to no-effect concentrations, but it should be kept in mind that the way of deriving of the values, including the use of safety factors may vary markedly. The intended level of protection is explained above.

Table A 16: Overview of some Water Quality Criteria (WQC) for heavy metals for high ecosystem integrity in freshwaters.

[1] Applicable for a hardness of 30 mg/l $CaCO_3$. Factors for calculating values at other hardness are presented in the guideline
[2] Applicable for hardness < 60 mg/l $CaCO_3$; at hardness > 180 mg/l $CaCO_3$: Copper 4 µg/l, Lead 7 µg/l and Nickel 150 µg/l; otherwise to be calculated
[3] Applicable for a hardness < 50 mg/l $CaCO_3$; at harness 50 to- <100 mg/l $CaCO_3$ 0.09 µg/l, 100 to <200 mg/l $CaCO_3$ 0.15 µg/l and ≥ 200 mg/l $CaCO_3$ 0.25 µg/l
[4] Bioavailable concentration
[5] Depending on hardness. At higher hardness higher concentrations
[6] Applicable for hardness from about 0.5 meq/l to 8 meq/l (= 25-400 mg/l $CaCO_3$)
[7] Arsenic V and chromium III to be converted to arsenic III and chromium VI
[8] Criteria are expressed in terms of the dissolved metals
[9] Applicable for hardness < 60 mg/l $CaCO_3$; at higher hardness higher concentrations

Guidelines	Aluminum in µg/l		Arsenic in µg/l	Cadmium in µg/l	Chromium in µg/l		Copper in µg/l	Lead in µg/l	Mercury in µg/l	Nickel in µg/l	Zinc in µg/l	Reference
	pH <6.5	pH >6.5			CR III	CR VI						
Australia/ New		27	1	0.06[1]	-	0.01	1.0[1]	1.0[1]	0.06	8[1]	2.4[1]	ANZECC/ARMCANZ (2000a)
Canada	5	100	5	0.09	8.9	1	2–4[2]	1–7[2]	0.026	25–150[2]	30	CCME (2016)
China	-	-	50	1	-	10	10	10	0.5	-	50	China (2002)
EU	-	-	-	0.08[3]	-		-	1.2[4]	-	4[4]		EC (2013a)
EU UK	-	-	50	-	4.7	3.4	1[5]	-	-	-	8[5]	UKTAG (2008c)
Japan	-	-	-	-	-	-	-	-	-	-	30	MoEJ (1997)
South-Africa	5	10	10	0.15–	12	7	0.3–	0.2–	0.04	-	2	DWAF (1996a,b)
UNECE	-	1.6	10[7]	0.07[6]	-	1[7]	2[6]	0.1[6]	0.003[6]	15[6]	45[6]	UNECE (1994)

US[8]	-	87	150	0.25[9]	74[9]	11[9]	-	2.5[9]	0.77[9]	52[9]	120[9]	US EPA (2016a)

Existing quality criteria and standards Heavy Metals (extreme Impairment)

The joint Australian and New Zealand Guidelines provide general trigger values for heavy metals. The presented values in »Table A 17 are the 80% level of protection of species. This level is associated with extremely disturbed systems. Even a 90% level of protection is referred to as “highly disturbed”.

The Canadian EQGs provide short-term exposure guideline values, but from the considered heavy metals a value is established only for cadmium.In China the surface water standards for most of the heavy metals are established for the five quality classes. The concentrations in »Table A 17 are the standards for the lowest class (grade V).

The EU WFD provides Maximum Acceptable Concentrations (MAC) for Cd and Hg at European Community level. These MAC-values are established to protect against short-term exposure.

In the South African WQGs the AEV is defined as the concentration above which there is expected to be significant probability of acute toxic effects to up to 5% of the species in the aquatic community.

The WQSs for the nine heavy metals for five classes are given by UNECE. The lowest quality class (V) means acutely toxic conditions in terms of concentrations, duration and frequency.

The US EPA presents WQC for acute and chronic exposure of aquatic life. The CMC (acute) as stated in »Table A 17 is an estimate of the highest concentration to which an aquatic community can be exposed briefly without resulting in an unacceptable effect and are intended to be protective of the vast majority of the aquatic communities in the US.

Table A 17: Overview of some WQC for heavy metals for extreme impairment of freshwater ecosystems.

[1] Applicable for a hardness of 30 mg/l $CaCO_3$. Factors for calculating values at other hardness are presented in the guideline

[2] Applicable for a hardness < 50 mg/l $CaCO_3$; at harness 50 to- <100 mg/l $CaCO_3$ 0.6 µg/l, 100 to <200 mg/l $CaCO_3$ 0.9 µg/l and ≥ 200 mg/l $CaCO_3$ 1.5 µg/l

[3] Depending on hardness. At higher hardness higher concentrations

[4] Applicable for hardness from about 0.5 meq/l to 8 meq/l (= 25-400 mg/l $CaCO_3$)

[5] Arsenic V and chromium III to be converted to arsenic III and chromium VI

[6] Criteria are expressed in terms of the dissolved metals

[7] Applicable for hardness < 60 mg/l $CaCO_3$; at higher hardness higher concentrations

	Aluminum in µg/l		Arsenic in µg/l	Cadmium in µg/l	Chromium in µg/l		Copper in µg/l	Lead in µg/l	Mercury in µg/l	Nickel in µg/l	Zinc in µg/l	Reference
	pH <6.5	pH >6.5			Cr III	Cr VI						

Australia/ New Zealand	-	150	94	0.8[1]	-	40	2.5[1]	9.4[1]	5.4	17[1]	31[1]	ANZECC/ARMCANZ (2000a)
Canada				1.0								CCME (2016)
China			100	10		100	1000	100	1	-	2000	China (2002)
EU				0.45[2]				14	0.07	34		EC (2013a)
EU-UK					32							UKTAG (2008c)
Japan			-	-	-	-	-	-	-	-	-	MoEJ (1997)
South-Africa	100	150	130	3[3]	340	200	1.6[3]	4[3]	1.7	-	36	DWAF (1996a,b)
UN ECE		75	360[5]	3.9[4]	-	6[5]	18[4]	82[4]	2.4	1400[4]	120[4]	UN ECE (1994)
US[6]	-	87	340	2.0[7]	570[7]	16[7]	-	6.5[7]	1.4[7]	470[7]	120[7]	US EPA (2016a)

In »Table A 17 an overview of heavy metal criteria in the considered WQGs is presented. All values are related to acute or short term effect concentrations. It is assumed that long-term exposure to concentrations of pollutants which may cause negative effects on short-term might be indicative for extreme impairment.

Proposed benchmarks for Heavy Metals (Category 1 and 4)
The presented overview in »Table A 18 highlights that existing guidelines provide for the high integrity status quality criteria for all heavy metals considered. The values in the guidelines are scientifically based and well documented. However, the criteria for setting standards can vary across guidelines. This variation is most likely due to the different procedures applied when deriving final limit values; most likely have been different toxicity data and different safety factors applied. Some differences moreover occur due to different corrections for hardness and the use of dissolved compared with total concentrations. All values presented in Table A 18 are intended to protect aquatic life for the long-term based on annual averages. Based on all values in the guidelines considered, benchmarks for heavy metals are proposed. The values are nearly always close to the median value and not the arithmetic mean. In most of the guidelines considered the criteria for some heavy metals depend on the hardness of the water. The proposed benchmark values in »Table A 18 are the proposed benchmarks for waters with low hardness. In many cases natural backgrounds are however lower than the proposed benchmarks. As recommended in some of the guidelines considered, estimating backgrounds for heavy metals may be a prerequisite because of the great variation between regions.

The presented overview in »Table A 18 highlights for the high impairment category that the guidelines considered provide criteria for acute or short term exposure levels. However, the guideline values found differ per guideline. This variation is most likely due to the different procedures applied when deriving final limit values. Based on the values in the guidelines considered, benchmarks are proposed for heavy metals which indicate extreme impairment of freshwater life. The proposed benchmark values are mostly close to the median value, not to the arithmetic mean of acute, short term and maximum allowable concentrations presented in the guidelines considered in this annex. The rationale is that acute and short-term guideline values may result in extreme impairment if exceeded continuously.

The guideline values for drinking-water (WHO, 2011) for some heavy metals are as follows: arsenic 10 µg/l, cadmium 3 µg/l, total chromium 50 µg/l, copper 2000 µg/l, lead 10 µg/l, inorganic mercury 6 µg/l and nickel 70 µg/l. For aluminum and zinc no drinking-water guideline values are established. The WHO drinking water guideline value for arsenic is the same as the proposed benchmark for high integrity (Category 1). The proposed benchmarks for extreme impairment for Cd, Cr, Pb, Hg, and Ni are in the same order of magnitude or somewhat lower as the WHO drinking-water guideline values in the WHO guideline, except for arsenic.

Table A 18: Proposed benchmarks for Heavy Metals
[1] Natural sources and geographical conditions may cause natural background values that differ from the benchmarks for high integrity. Instead of these benchmark values natural background concentrations may be used for setting criteria for high integrity.
[1] Applicable for waters with low hardness (< 60 mg/l $CaCO_3$). In case of higher hardness the values may be higher.

	High Integrity (Cat 1) [1]	Extreme Impairment (Cat IV)
Aluminum (µg/l) pH <6.5 pH >6.5	 5 10	 - 100
Arsenic (µg/l)	10	150
Cadmium (µg/l) [2]	0.08	1.0
Chromium (µg/l) [2] Cr III Cr VI	 10 1	 75 40
Copper (µg/l) [2]	1	2.5
Lead (µg/l) [2]	2	5
Mercury (µg/l) [2]	0.05	1.0
Nickel (µg/l) [2]	20	50
Zinc (µg/l) [2]	8	50

Annex 3

Setting benchmarks for ecosystem health: an example from the United States

The scientific methods and inferences for developing benchmarks differ depending on whether the assessment endpoint is for an exposure (e.g. chemical or physical water quality criteria) or for a desired biological condition such as the diversity and abundance of aquatic life (biocriteria). The specific methods used for each vary globally. This annex provides an example from the United States and includes a discussion of useful scientific elements of Water Quality Benchmarks (WQBs), the importance of setting baselines for comparison during development and implementation; and some of the more common methods for deriving laboratory-, field-, and model-based benchmarks. Additionally, the value and use of descriptive models for interpreting results across differing programs, and for communicating effectively with the public and policy makers about the current status of aquatic resources and their potential for recovery to a more desired condition is discussed.

Useful scientific elements for benchmarks

Policies for effective use of benchmarks as protection or restoration goals depend on scientific information that consists of more than the concentration of a pollutant or a minimal number of biological taxa. The scientific process of benchmark development is described in environmental assessments (see Section 2.5 on ecological risk assessment) and typically includes:

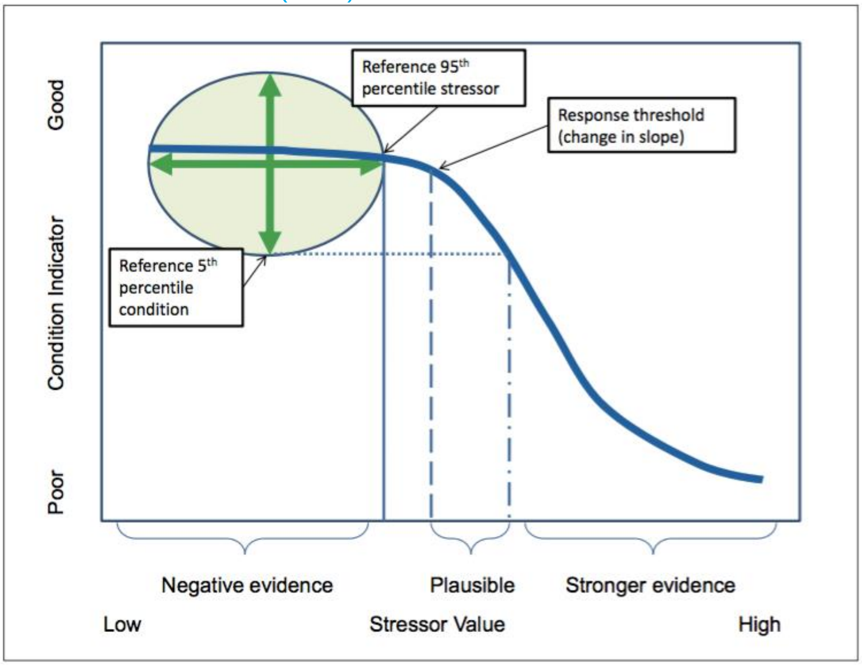
- The resource attribute to be protected (e.g. water quantity and quality, biological diversity and biological resource abundance);
- The relevant water body types and geographical limits (e.g. streams, estuaries, springs, lakes);
- The societal value of the water or biological resource, sometimes referred to as a designated use (e.g. recreation, water supply, aquatic life, cultural value);
- The narrative and/or numeric description of the benchmark (e.g. exposure concentration, duration, and frequency; invertebrate index thresholds);
- Implementation considerations (e.g. natural parameters such as water hardness and season, mixing zones, variances); and
- Baseline characterizations to avoid back-sliding and protect exceptional natural resources.

Comparison of biologically-based benchmarks and stressor-based benchmarks

WQBs are typically sorted into biologically-based and stressor-based benchmarks, because the methods for developing them are very different. Biologically-based benchmarks use the presence, abundance or other attributes of biological taxa to assess ecosystem condition. In the scientific literature, one may see biologically-based benchmarks referred to as biological criteria or biocriteria and stressor-based benchmarks as Water Quality Criteria (WQC) when they are used for regulatory purposes. Benchmarks that capture excess concentration, frequency, and duration of exposures to physical (e.g., temperature), chemical (e.g., metals, organics), and biological (e.g., pathogens, non-native or invasive species) stressors in aquatic systems are referred to as stressor-based benchmarks, criteria, guidelines, or standards, depending on usage. Both are developed through a predictive assessment process (Cormier and Suter 2008; US EPA 2010b). Biologically-based benchmarks are typically developed from field observations, however, modeling is increasingly being used to help set expectations (e.g., random forest modeling; RIVPACS (Wright 1995). WQB may be developed from field or laboratory data, models based on principles of physics or chemistry, or a combination of these information sources.

A WQB assessment differs from a conventional risk assessment in that an acceptable effect endpoint is used to predict either an effect (biologically-based benchmark) or a threshold for an exposure (stressor-based benchmark). The stressor-response model (»Figure A 1) is the same in conventional risk and benchmark assessments, and for most physical and chemical WQBassessments in the US, the effect endpoint is the 5th percentile of affected taxa. In a biologically-based benchmark assessment, the effect endpoint is a statistical threshold discriminating metrics (or observed /expected model values) of the biological composition at reference condition (RC) from non-reference condition sites or multiple thresholds along a gradient of stress.

Figure A 1: Conceptual relationship between stressor dose and gradient of biological responses .
Source: Gerritsen et al. (2010).



For biologically-based benchmarks, the exposure portion of the stressor-response model is the set of stressors associated with a set of sites. The sites may be defined as categories (e.g., reference /non-reference) or along a gradient of exposure. The reference category and disturbance gradient are set relative to conditions of the water resource in the region. For example, exceptional resources may have a standard of protection that protects all of the most sensitive, native species; whereas, thresholds for areas with moderate or extensive human disturbance may allow for loss of some sensitive species and increase in proportion of species able to tolerate increased levels of stressors. Threshold selection for biologically-based benchmarks recognizes that ideal conditions are not attainable in all places because of unalterable modifications that have been made in the landscape over the course of human settlement (e.g., channelized streams, widespread agriculture, urban development, etc.). The stressor-response model characterizes the range of current conditions relative to disturbances so that differences among sites and achievable goals can be recognized.

WQBs are used to assess the condition of water bodies and may be specific to a water body type (e.g., rivers, lakes, wetlands) or water bodies of a region. These condition assessments may be used in setting regulatory expectations of condition and may result in a designation of acceptable or unacceptable condition (Cormier and Suter 2008; US EPA 2010a,b). When stressor-based benchmarks are not met, the stressor of concern must be treated or remediated. However, when a biologically-based benchmark is not met, and the cause is not known, a causal assessment is performed to determine what needs to be remedied in order to restore the biota typical of better conditions (Norton et al 2015, US EPA 2010a,b).

The impetus for management action varies from country to country. The European Union’s Water Framework Directive (EU WFD) distinguishes “Good Chemical Status” from “Good Ecological Status”, and water bodies should achieve both (“one out - all out”).

Similarly, in the US, remedial action is triggered when WQBs or biologically-based benchmarks are not met, or if toxicity is identified from effluents or ambient waters (US EPA, 2014a). Independent applicability of benchmarks was chosen as a policy because each benchmark measures different aspects of environmental quality and may not measure a specific type of stressor or effect. For example, a biologically-based benchmark may not detect a problem when a method only samples riffles and there is only one riffle in a kilometer of stream, whereas the rest of the stream may exhibit poor quality due to excess sediment from erosion. Alternatively, laboratory-based benchmarks may be met, but not biologically-based benchmarks, which take into account sensitive taxa that were not tested using laboratory-based benchmarks (US EPA, 2013b). These differences in the chosen detection benchmarks can be quite large. For example, in a 1990 study of stream in the State of Ohio, US, biologically-based benchmarks were not met in 49.8% of the cases where chemical benchmarks were met (e.g., Yoder and Rankin 1996). In contrast to decision-making using independent applicability, some advocate action only when all three endpoints are not met, such as within the sediment quality triad (Chapman 1990, 1995). However, using the sediment quality triad method requires exceptionally well-established benchmarks and high quality and abundant spatial and temporal site data.

Stressor-based water quality benchmarks

The development of stressor-based benchmarks follows the basic scientific assessment process that includes describing the problem, performing analyses, and forming conclusions that culminate in the description of a benchmark that may be adopted as a regulatory standard or remediation goal (Cormier and Suter 2008; Suter and Cormier 2008; US EPA 1998). These three steps are described below for developing benchmarks using laboratory-, field-, and model-based approaches. The section on analysis is divided into subsections for each approach in order to point out differences and special considerations.

Problem Formulation

Problem formulation describes the objectives of the stressor-based WQB assessment and its assessment endpoints, measurements, and methods. At a minimum, a problem formulation includes a description of the stressor and factors that may alter exposure, how the stressor is measured, and why it is considered a problem. For example, aluminium is toxic to aquatic organisms, and its bio-availability is affected by pH and hardness. The problem formulation also includes what is known about the stressor's sources, mode and mechanism of action, and how it affects the biota or ecological functions that led to the selection of the assessment endpoints. These types of information form an explanation for the selection of a measure of effect and rationale for the selection of the analytical method used to derive the WQB.

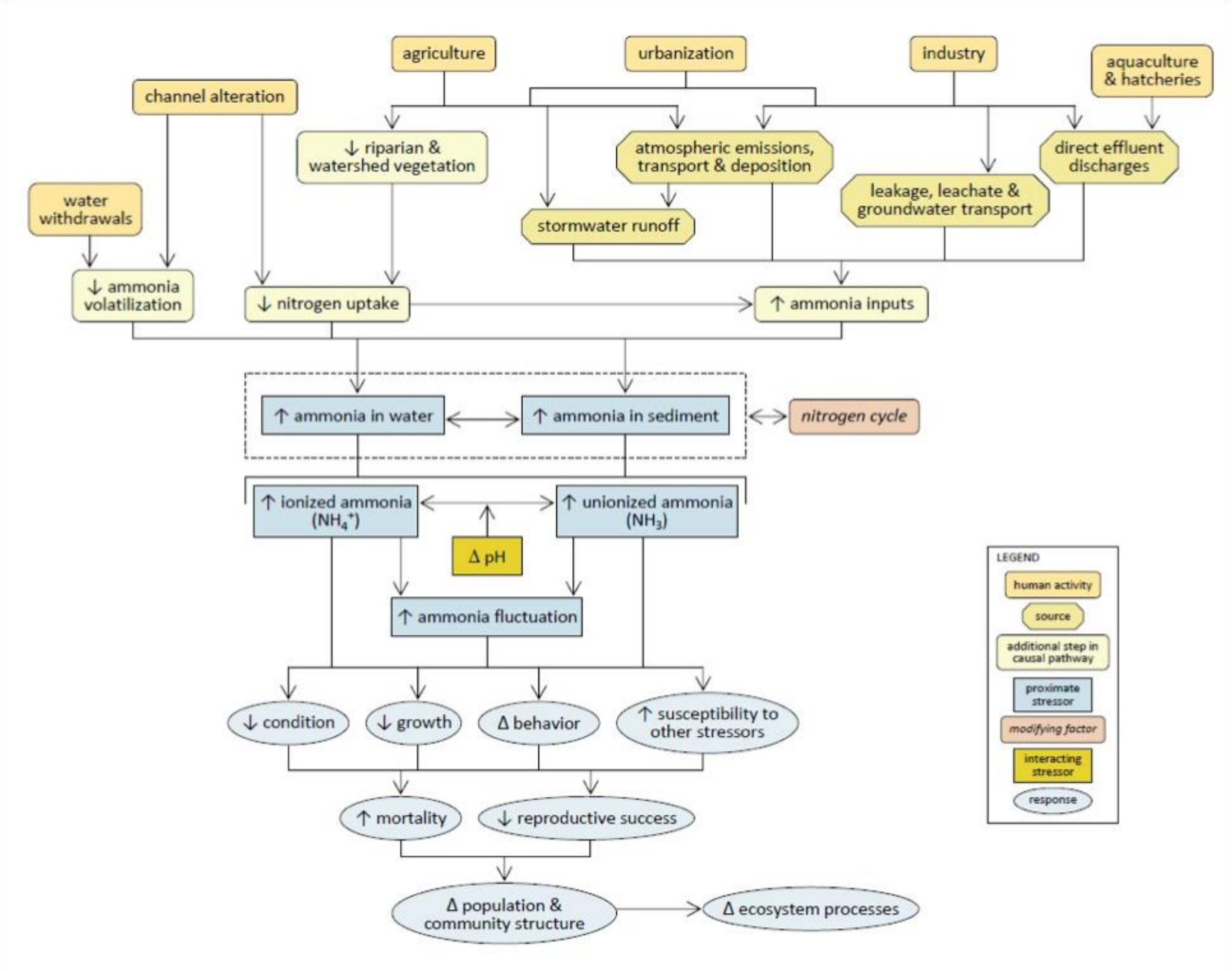
Problem formulation also addresses the categories of exposure for which benchmarks will be derived. Commonly, the categories are "acute" and "chronic". Acute benchmarks address exposures of short duration such as those from spills, accidents, pesticide applications, or emissions from batch industrial processes. Acute benchmarks are typically based on lethality because it is assumed that populations can tolerate short periods of sublethal effects. Chronic benchmarks address in contrast longer-duration exposures such as from routine emissions or frequent episodes. They include sublethal effects such as reduced growth and reproduction as well as lethality due to cumulative effects of long-duration continuous or frequent episodic events. Data and methods are not always the same for these two categories of benchmarks. For example, in the US, acute WQC are based on 48 to 96 hour lethality in toxicity tests, whereas lethality, growth and reproduction are used for chronic criteria. EU EQSs estimate two values, an annual average and a maximum allowable concentration (EC, 2013a).

Most benchmarks are based on concentrations in water or other media. However, media concentrations of chemicals that act through dietary bioaccumulation may be poorly related to effects. For example, the selenium (Se) WQC in the US is based on tissue concentrations because the sensitive effect is deformities of human embryos due to Se transferred from maternally accumulated selenium (US EPA, 2014b).

Field-based or mesocosm-based data might be selected because the mode of action is not measurable in a small scale (laboratory) experimental design. For example, benchmark values for major ion concentrations have been developed using field data and confirmed in artificial stream mesocosm studies (US EPA, 2011b,c; Nietch, 2014; Clements, 2014). A diagram showing the linkages among sources, stressors, exposure pathways, mechanisms and modes of action, and lethal and sublethal organismal and population level effects can be a useful communication tool for conveying this information (see »Figure A 2; US EPA, 2010b,c; Norton et al., 2015). These conceptual models may reveal steps in the causal pathways that are then more readily able to be controlled with best-management practices so that the protective benchmark can be met.

Most WQB are distinct for freshwater and marine systems. The applicability of benchmarks may be independent of geography and climate as in the case of most conventional chemical contaminants such as metals and organic compounds. In contrast, field-based WQB are customized for certain geophysical or biogeographical regions. For example, nutrient criteria in the US are derived for geophysically similar regions (US EPA, 2000a,d).

Figure A 2: A conceptual model for ammonia used in the US EPA Water Quality Criterion (WGC) for ammonia. Source: US EPA (2013a) and http://www.epa.gov/caddis/ssr_amm_int.html.



Analysis

Development of numeric stressor-based WQB is based on three assumptions. First, causal relationships exist between stressors and environmental effects. Second, these causal relationships can be quantitatively modelled. Finally, when exposure to the stressor remains within a defined range predicted to be protective by the quantitative model, adverse effects will not occur, and desired uses will be safeguarded. Therefore, a benchmark must be based on a reasonably consistent and scientifically defensible stressor-response relationship. These relationships are characterized using associations from laboratory experiments, field observations, or mechanistic models.
Important considerations for developing WQBs include sources of information and data, rationale or analysis of regional applicability, and methods to actually derive the benchmark. These should include a description and results for the process for the selecting and determining the suitability of the data as well as quality assurance and quality control procedures used during database construction and analysis. In a next step, a stressor-response association is modelled and a hazardous concentration calculated. Analytical methods vary but should include a characterization of uncertainty for the model.

Introduction to approaches

The United States Environmental Protection Agency (US EPA), the European Commission (EC), and others allow the use of laboratory-, field-, and model-based approaches (e.g., EC, 2000b). The default method is laboratory-based. It has the advantage of inference based on experimentation; the causal relationship can be manipulated, exposures randomized, results replicated, and other influences minimized or controlled. Furthermore, methods have been standardized and can be used to evaluate stressors before they are released into the environment.

In some situations, however laboratory tests cannot replicate the full range of ambient exposures, effects, or interactions. Some stressors and effects are impossible to be studied in a laboratory setting. For example, tests of large species are logistically prohibitive and endangered species are protected from routine testing. Migration, predation, and other behaviours, and some life stages, such as spawning, are seldom investigated by standard laboratory procedures. Susceptible species and sensitive life stages may be difficult to maintain and test in the laboratory. Effects that involve interactions among species are difficult to study. Latent effects due to an earlier exposure are not measured (e.g. reduced adult reproductive success owing to exposure during a critical stage of embryogenesis). In addition, the relative sensitivity of most species is not known ahead of time, and it is impractical to test a substantial fraction of the species inhabiting an ecosystem. Complex exposure pathways and bioaccumulative chemicals are not readily tested and some exposures are impractical to replicate, such as highly variable concentrations and interactions within mixtures and with the environment. A potential solution is to use field observations instead of laboratory toxicity tests.

Approaches using field data have advantageous properties (Posthuma et al 2002, Struijis et al 2010; Cormier, et al. 2013). Unlike toxicity tests and models, field observations document the changes that are actually occurring in the environment. Field studies directly measure exposures of entire communities and include direct and indirect effects across entire life cycles. Furthermore, unlike chronic test endpoints, which are difficult to interpret, field responses such as population removal are understood by most people. However, field-based approaches also have disadvantages. Very large sets of paired stressor and biological response data are necessary. The data set must include effects of many sensitive taxa, and must include a range of exposures sufficient to cause a full range of effects. Also, exposures and deleterious effects must already occur in the environment at harmful levels. This is not the preferred situation. Another difficulty is the co-occurrence of other stressors that may confound a field-based stressor-response model. A confounder is an agent that co-occurs with the stressor of interest and that may distort the empirically modelled stressor-response association. Potential confounding can be evaluated and minimized, but that requires data that may not be available, and in some cases, the potential for confounding remains uncertain (Suter and Cormier 2013).

When there are sufficient data and a deep understanding of the interacting factors, a WQB can be based on mechanistic models. Stressor-response models may rely heavily on principles of physics, chemistry, and biology. The potential advantages of these models include speed, flexibility, and inclusion of mechanisms at different levels of organization. At present, the only example of a model-based benchmark is the WQC for copper which uses the Biotic Ligand Model (BLM; DiToro et al 2001; US EPA 2007) which examines the bioavailability of metals in the aquatic environment and the affinity of these metals to accumulate on the gill surfaces of organisms. Other promising models for non-conventional stressors such as ionic mixtures or stream sediments are primarily useful for benchmarks that are based on habitat quality rather than biological condition and may use a deviation from background or RC (Olson and Hawkins 2012; Kaufmann et al 2009). Ecosystem and water quality models may become more valuable in the future especially for predicting complex ecological responses and effects of mixtures.

Although it is conceivable that all three approaches could be used and results compared, in practice nearly all WQC in the United States and elsewhere have been derived using standard laboratory toxicity tests. Development of field-based benchmarks has lagged behind because suitable large data sets have only recently begun to be available, and field experiments are more costly than laboratory experiments. Ultimately, professional judgement is needed to choose whether to use a laboratory-, field-, or model-based method to develop WQBs. For field methods, a good practice is to determine whether there are suitable data and whether the exposure-effect model is reasonably free of confounding by other environmental parameters. For laboratory-based methods, a good practice is to determine if data are available for sensitive taxa and whether the assessment endpoint is relevant to effects that occur in the field. If neither method is suitable, it may be necessary to collect additional data or develop sensitive species or mesocosm tests.

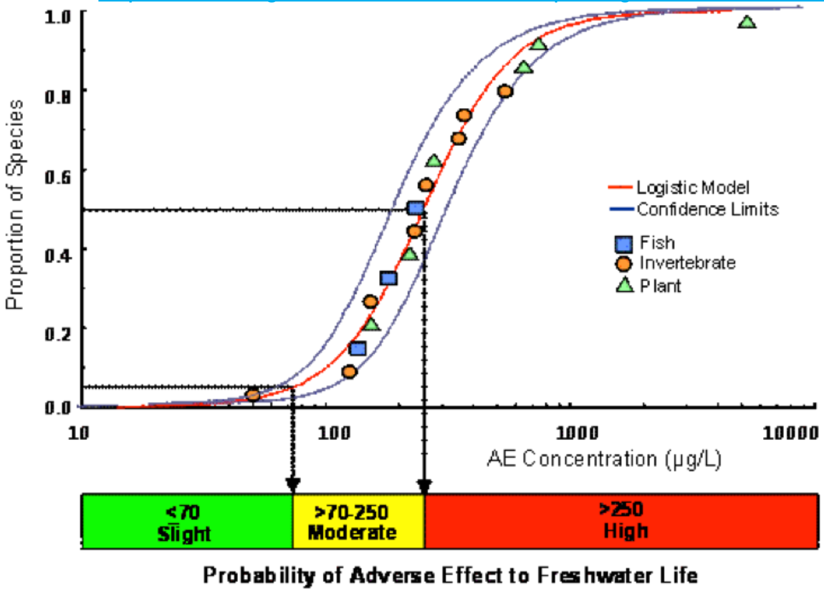
Laboratory-based approaches

Laboratory-based benchmark development in the US has relied almost entirely on one scientifically rigorous method (Stephen et al. 1985). The method characterizes the stressor-response relationships using results from toxicity tests of species in at least 8 different families of taxa. However, as data for more species have accumulated over the last 30 years (see ECOTOX, US EPA 2015a), usually more genera are represented in the stressor-response curve from which the benchmark is estimated. In some cases, field data have revealed that some species or taxonomic groups are not protected by an existing benchmark, and test methods using sensitive species are developed and the benchmark is updated (e.g., Ammonia Criteria, US EPA, 2013a). In the United States and Canada, acute benchmarks are derived from half the 5th percentile Hazardous Concentration (HC05) from Species Sensitivity Distributions (SSDs). Chronic criteria for chemicals with chronic values for the 8 families are 5th percentiles of the chronic SSD. However, most chronic criteria are derived from the acute SSD and an acute/chronic ratio. Thus, most species are protected most of the time. The SSD represents how species in general are expected to respond in an exposed community based on a set of representative genera (»Figure A 3) (Posthuma et al 2002).

A compilation of US EPA's national recommended WQC for the protection of aquatic life and human health in surface water is available for approximately 150 pollutants (http://water.epa.gov/scitech/swguidance/standards/criteria/current/index.cfm). Canadian Environmental Quality Guidelines, equivalent to benchmarks, can be accessed from http://st-ts.ccme.ca/en/index.html.

In the EU, Directive 2008/105/EC (the Priority Substances Directive established a list of 33 priority substances and 8 other Environmental Quality Standards (EQS)s and is continually expanded. A yearly average and a Maximum Allowable Concentration (MAC) for short term pollution peaks are provided. A table of priority compounds are listed at: http://ec.europa.eu/environment/water/water-framework/priority_substances.htm

Figure A 3: Species sensitivity distribution (SSD) for freshwater toxicity data for the alcohol ethoxylate homologue C13.7 EO5 and associated effect levels for freshwater life. Each geometric shape represents a laboratory effect for a particular species. In this example, acute and chronic effect endpoints are shown. The Canadian Federal Water Quality Guideline (WQG) at the 5th percentile is 70 µg/L for alcohol ethoxylate homologue C13.7 EO5.
Source: http://www.ec.gc.ca/ese-ees/default.asp?lang=En&n=164786DB-1

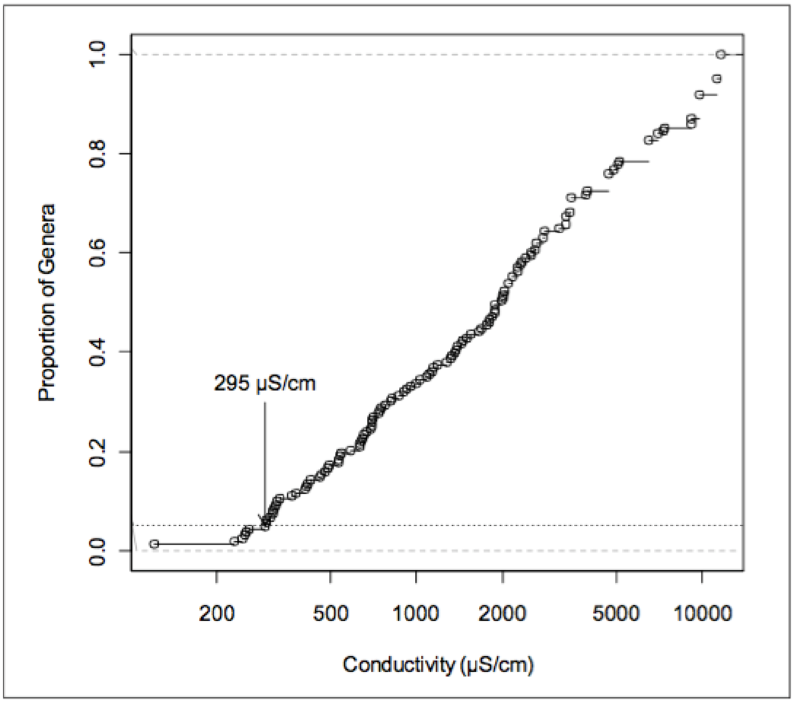


Field-based approaches
The option of using field data is permitted by the EU WFD (European Parliament and Council, 2009) and has been recommended by the US EPA) Science Advisory Board (US EPA 2011b). It has been specifically recommended for ionic mixtures, suspended and benthic sediment, and nutrient benchmarks (Cormier et al., 2008, US EPA 2006a).

Field-based methods vary depending on the effects observed in the field and the types of data that are available. One method uses a SSD to derive a field-based benchmark at the 5th percentile (e.g., Struijis et al., 2010, US EPA, 2011b,c). In contrast to the laboratory-based SSD which typically uses median lethal concentrations (LC50s) or other laboratory test endpoints, the effect measurement in the field-based method uses either taxa optima or tolerance limits estimated from field monitoring. For example, the concentration of a stressor at the 95th percentile of the observed occurrence of a genus can be estimated from stream monitoring data. This is referred to as an extirpation concentration, the ambient concentration beyond which a taxon is rarely observed in its original habitat or range. These XC95 values have been used to develop a benchmark for a mixture of dissolved ions (US EPA, 2011b,c) (»Figure A 4). SSD methods using other effect endpoints have been developed for nutrient benchmarks (e.g. Struijs et al., 2010).

Other derivation methods use best professional judgment or statistical thresholds which may be difficult to link to a biologically meaningful effect threshold (US EPA, 2010a,b). In Canada and the US, field-based methods were used to develop benchmarks for suspended and deposited sediment and may use a weight-of-evidence approach to select a provisional benchmark (Benoy et al., 2012). In some cases a stressor-response model is not used. For example, 10% greater than background is recommended for suspended sediment in British Columbia, Canada (CCME, 2016) and 25% of background for nutrients in the US. (US EPA, 2000a). The US EPA has also released guidance for developing WQB for stressors with physical and biological modes of action (US EPA, 2000a, 2006a, 2010a,b, 2011b,c). The US states of Connecticut and Maine have developed a stressor-response model for mixed stressors associated with impervious cover (Bellucci et al. 2008; MDEP 2012). Because field-based methods are not standardized and method development is an area of active research, careful consideration of options are needed before selecting a field-based method.

Figure A 4: Example of a species sensitivity distribution (SSD) depicting the proportion of genera extirpated with increasing ionic strength measured as specific conductance. Each point on the SSD plot represents an XC95 value of one of the 163 genera arranged from the most to the least sensitive. The 5th percentile hazardous concentration (HC05) used to define the benchmark is the 5th percentile of the SSD (dotted horizontal line) and occurs at 295 µS/cm.
Source: West Virginia Department of Environmental Protection. Graphs from US EPA (2011a).



Model-based approaches

As previously mentioned, model-based benchmarks have rarely been adopted for use (Di Toro et al., 2000, Santore et al 2000). The biotic BLM uses several water chemistry parameters to calculate a freshwater copper benchmark (US EPA 2007) but other metals have also been modelled (Santore et al., 2000). The model is useful because the 1986 laboratory-based benchmark for copper was not protective at low pH and overly protective at higher Dissolved Organic Carbon (DOC) levels and did not take into account other factors such as major ions and alkalinity. Furthermore, the method allows the WQB to be tailored to site-specific water parameters. Three types of chemical interactions are modelled: metal-inorganic ligand interactions (Santore and Driscoll 1995), metal-organic matter interactions (Tipping 1994), and biotic ligand interactions (Playle et al., 1993a,b). This exposure model is then combined with a physiological gill site (i.e. gill surface) interaction model (Pagenkopf 1983). Toxicity test and other information are used to calibrate the model.

Assessment Conclusions

The conclusions of a criterion assessment characterize the chronic and acute benchmarks and specify the duration and frequency of exposures. For particularly sensitive and important species, e.g., endangered species, a special benchmark may or may not be provided. Tables of the effect levels used in any kind of an SSD are included along with literature citations, when appropriate. Any seasonal, geographic, natural history, chemical, or physical factors that may affect the benchmark are discussed. Details of analyses and implementation tools, such as calculators used when ambient factors affect the benchmark, may be included in appendices. Also, the results are summarized for supporting analyses, such as validation of the model, assessments of causation or potential confounding of the model.

Biologically-based benchmarks

Biologically-based benchmarks have the advantage of detecting effects of many stressors that are not measured by existing WQB, such as pathogens, invasive species, habitat alteration, and flow alteration. They also incorporate effects that are not measured in standard laboratory tests but can alter populations and communities such as endocrine disruption, immune impairment, and low fecundity. As a result, it is not uncommon for biologically-based benchmarks to detect a high percentage of poor conditions in a region due to habitat modification (Yoder and Rankin 1996).

Unlike a stressor-based benchmark, the exposure measurement of biologically-based benchmarks deliberately represents multiple stressors (Karr and Chu 1999). Biologically-based benchmarks that use an index are well adapted for discriminating at least three categories of biological condition “good”, “fair”, and “poor,” but the biological effects are not evident from the index score alone.

In order to understand the effect, it is necessary to examine the underlying metric scores or attributes of affected species. To maximize discriminatory power beyond the above mentioned 3 condition classifications, metrics must be rigorously chosen for those that avoid redundancy and that have a broad range of possible values. For example, for a metric where the number of species is summed, it is mathematically impossible for a metric with a potential range from 0-3 species to distinguish five condition classes. Furthermore, because the exposure parameter of a typical index consists of multiple stressors, the cause is not defined and a causal assessment is needed for taking management action (Norton et al., 2015; US EPA, 2010b,c).

The methods for the selection of an effect endpoint vary by developer and approach and are described in greater detail in the following analysis section. Unlike stressor-based benchmarks that use a defined biological effect level to calculate a stressor intensity from the stressor response model (e.g., 5th percentile of affected taxa), biologically-based benchmarks use statistical tests to identify thresholds on the biological axis, such as an index score that discriminates thresholds along a gradient of increasing human disturbance or reference sites from categories of affected sites. Biologically-based benchmarks are typically developed from field observations using the RC approach but data and outputs from models and historical records may be used. Biologically-based benchmarks may be used to establish multiple thresholds along a gradient of stress. A combination of data sources to establish biologically-based benchmarks is particularly relevant for waters in highly disturbed regions of the country where RCs may not exist and the impact of human activity on aquatic resources has a long history (legacy effects), widespread and pervasive.

Most commonly, a biologically-based benchmark is used to evaluate environmental condition. However, the stressor-response relationships developed with a biologically-based benchmark are also the technical basis for various types of policy decisions, including identifying and analysing environmental problems, setting goals and priorities, managing programmes, and monitoring for results which are well established in the US, Canada, European Union, Australia, New Zealand, South Africa and other countries (Cormier and Messer 2004).

Problem Formulation

Problem formulation describes the background information and rationales for the selection of assessment endpoints, measurements, and methods and the geographic and water body type. Rationales are described for selecting parameters making up the exposure axis of the model or the parameters and thresholds that were used to assign reference or non-reference status (see example »Table A 19). These decisions are not made a priori and the actual selection of reference sites occurs after analysis, with only the rationales identified during problem formulation. For example, in West Virginia, US, least disturbed reference sites are selected based on 11 criteria (see Table A 19). Criteria for Human Disturbance Gradient (HDG) (Hawkins, 2006; Davy-Bowker et al., 2006) and reference selection often include land use characterizations, such as >90% of land use in native vegetation. Similarly, rationales and methods for selecting ecological and biological metrics are described, but actual selection occurs during analysis. Sampling design, data sources, and quality assurance processes are described. Land use characterization methods, and laboratory and field measurement methods are provided.

A flow diagram of the analytical process is a very useful tool for communicating the logic supporting each analytical step and the many judgments that are made when developing biologically-based benchmarks. Although it is not common practice, a conceptual model linking metrics to sources and stressors can make it clear why the selected metrics are ecologically relevant and the range of stressors and effects that are potentially measured by the biologically-based benchmark. It leads to a way to begin to interpret the cause of the condition measured at a particular location (US EPA, 2010b,c; Norton et al 2015).

Table A 19: Reference criteria for West Virginia 1996-1997 Stream Assessment.
WV = West Virginia; WAP= Watershed Assessment Programme (West Virginia);
OWR = Office of Water Resources (West Virginia); BPJ = Best professional judgement.
Source: Modified from Gerritsen et al. (2000).

	Parameter	Criterion	Explanation
1	Dissolved oxygen	6.0 mg/l	Taken from “WVWater Quality Standards” as developed by the State Water Resources Board (SWRB)
2	pH	6.0 and 9.0	Conductivity and pH are based on observations of WAP and OWR data and from BPJ of experienced OWR field personnel
3	Conductivity	<500 µmhos/cm	
4	Fecal coliform	<800 colonies/100 ml	This limit is double the maximum set by the SWRB (where the standard is no more than 400 colonies/100 ml in more than 10% of all samples taken during the month. Fecal levels exceeding the standard in some streams occur where no human impacts were possible (possibly due to wildlife populations).
5-8	Epifaunal substrate; Channel alteration; Sediment deposition; Bank disruptive pressure score	≥11	Criteria 6-11 are adapted from Rapid Bioassessment Protocol (RBP) habitat assessment modified for use in the US EPA Environmental Monitoring and Assessment Program (EMAP). WVWAP sampling strategy occurs at or near the mouths of streams which can negatively bias this score, so this score is relaxed. Otherwise, few streams (if any) would meet the selection criteria.
9	Riparian vegetation zone width score	≥ 6	Variable depending on watershed
10	Total habitat score	65% of maximum of 240	% is variable depending on watershed
11	Evaluation of anthropogenic activities and disturbances	Best professional judgement	No obvious sources of non-point-source pollution ; No known point source discharges upstream of assessment site

Analysis

Selection of methods

Methods for developing biologically-based benchmarks have advanced since the advent of larger monitoring databases and greater computing power for analysis. Sampling methods and designs have been evaluated for various applications (Cormier and Messer 2004). Best professional judgments are backed by ecological data analysis. Analytical methods have advanced the discriminatory power of indices. However, even when data sets are relatively small, at least three levels of biological condition can be distinguished and over time, more data allows the stressor-response model to be revised and improved (see US EPA, 2011c, in there section 2.1, for discussions on program rigor; see also

»Figure A 5 and » below). This section describes the general approaches in order of increasing sophistication.

Characterizing Exposure
Criteria for defining reference and non-reference sites, or HDG tiers are selected prior to analysis and rely on best professional judgment. Typical considerations are water and habitat quality, proportion of natural vegetation, and minimal human disturbance and sources of point and non-point pollution. For the HDG, the tiers are defined from nearly pristine conditions through various lesser quality conditions. In highly modified landscapes, tier one and two may not occur, such that the HDG has a more constrained range. The HDG and reference/non-reference characterization is equivalent to an exposure characterization in risk assessment and represents multiple stressors

Characterizing stressor-response
There are at least three common approaches for developing a stressor-response model for biologically-based benchmarks. First, the taxa observed at a site can be compared to the taxa expected to occur at reference sites or a classification of the HDG. Second, individual metrics can be assessed for their ability to respond and distinguish among different stressors and these metrics are then combined into an index (e.g. multimetric index). These metrics may be modelled to account for natural variation, such that site-specific expectations can be compared to observed values (e.g., random forest modeling). The difference between the expected and observed values for each metric are then standardized to include in a multimetric index similar to unmodeled metrics. A third approach, albeit less common, is to define critical attributes and functions of an ecosystem, develop measures for those attributes and functions, and then describe how each attribute changes in response to stress. The Biological Condition Gradient (BCG) (Davies and Jackson 2006) is a good example of this approach. This approach is used to interpret biological condition along a standardized gradient, regardless of assessment method. It is therefore, a useful tool for summarizing data across programs that use different approaches or are at different stages of development. The section on developing narrative criteria discusses the BCG approach in greater detail.

Identifying the benchmark
There is no standard method for selecting an ecologically meaningful effect endpoint. Best professional judgment, precedence, a proportional difference from background or Reference Condition (RC), and statistical ability to discriminate one category from another have been used. However, as soon as the endpoint is selected, it is used with the stressor-response model to identify the biological conditions that meet the biological quality objective. For a multimetric index, such as an Index of Biotic Integrity (IBI), the scores may define qualities such as exceptional, good, fair, and poor. For approaches using a BCG, a biologically-based benchmark is defined as meeting tiers of greater or less ecological integrity. The observed-compared-to-expected type of biologically-based benchmark indicates the degree of difference from RC. A designation of impairment is ultimately a policy matter and not a technical one. As mentioned previously, the stressor-response model provides technical details that can guide management protection and remediation.

Assessment conclusions
The conclusions of a biologically-based benchmark assessment characterize thresholds between the RC and one or more or a range of disturbed conditions. Disturbance is identified based e.g. on natural vegetation, chemical and physical ecosystem condition, and sources of pollution. In addition to the development of the benchmark, the stressor-response model is valuable because it describes the full range of potential conditions. Any seasonal, geographic, natural history, chemical, or physical factors such as seasonal index period that may affect the benchmark are described. Details of analyses, individual stressor-response profiles for individual metrics, and implementation tools, such as calculators used when ambient factors affect the benchmark, may be included in appendices. Distinct benchmarks for special designation, such as exceptional waters, are identified. Also, the results are summarized for supporting analyses, such as validation of the model.

Figure A 5: Diagram illustrating incremental improvement of biologically-based benchmarks. Source: Davies and Jackson (2011).

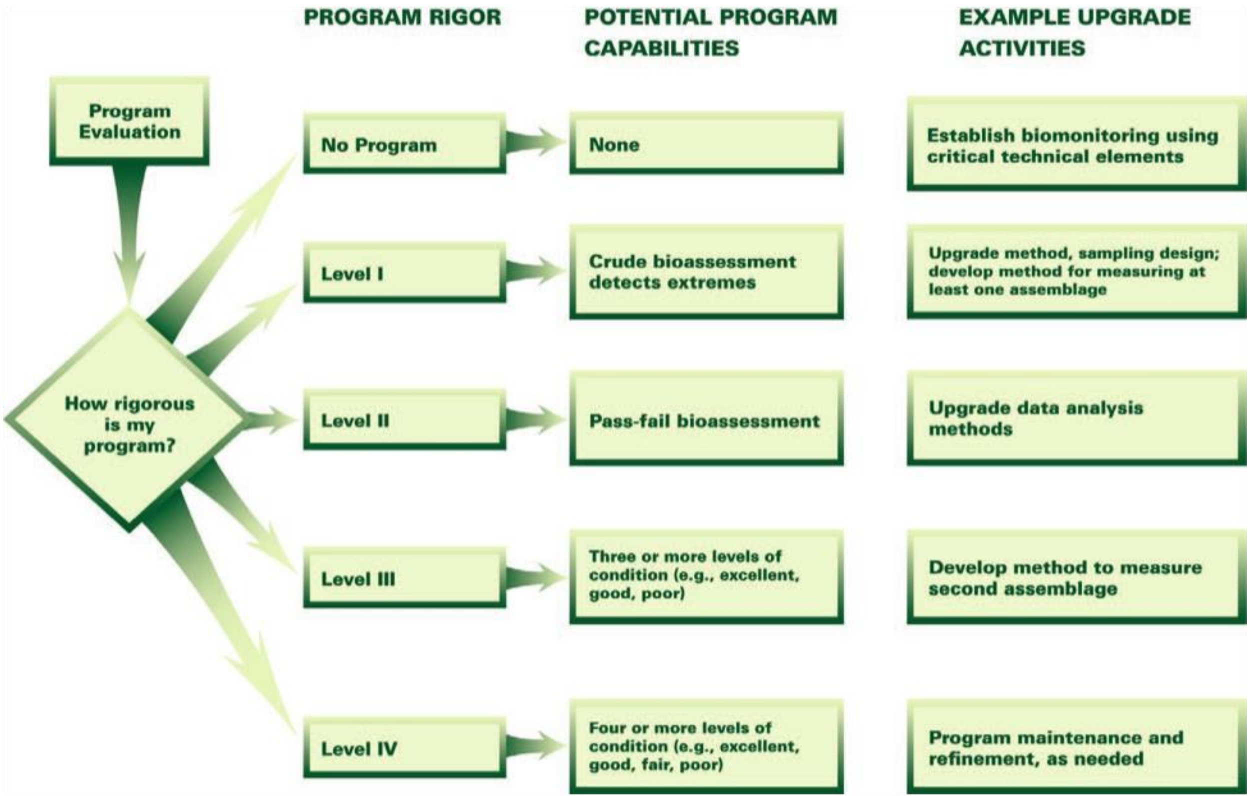


Table A 1: Key features of the technical attributes for levels of rigor in state/tribal biological assessment programs (streams and rivers). This template can be modified and applied to other water body types.
Source: Davies and Jackson (2011).

	Attributes of levels of biological assessment program rigor			
Key features	**Level 1**	**Level 2**	**Level 3**	**Level 4**
Temporal and spatial coverage	Variable data collection times; upstream/ downstream and fixed stations	Index period for convenience; non-random design at a coarse scale	Calibrated seasonal index periods; statewide spatial design using rotating basins at a coarse scale	Scientifically-derived temporal sampling for management decisions; multiple spatial designs for multiple issues
Natural classification of aquatic ecosystems	No partitioning of natural variability; no incorporation of differences in stream characteristics such as size, gradient	Classification usually a geo-graphical or other similar organization (e.g., fishery- based cold or warmwater; lacks intra-regional strata [size, gradient])	Classification based on a combination of landscape features and physical habitat structure; considers all intra- regional strata and specific ecosystems	Fully partitioned and stratified classification scheme that transcends jurisdictions and recognizes zoogeographical aspects of assemblages
Reference conditions	No reference conditions; presence and absence of key taxa are based on best professional judgment	A site-specific control or paired watershed approach can be used for assessment; regional reference sites are lacking	Reference conditions used in watershed assessments; regional reference sites are too few in number or spatial density	Regional reference conditions are established in the applicable water body ecotypes and aquatic resource classes
Sampling and sample processing	Approach is cursory, relies on operator skill and best professional	Textbook methods are used rather than in-house	Methods are calibrated for state purposes	Same as Level 3, but methods cover multiple assemblages;

	judgment, producing highly variable and less comparable results	development of standard operating procedures to specify methods	and are detailed and well documented; supported by in-house testing and development	high taxonomic resolution
Data management	Sampling event data are organized in a series of spreadsheets	Separate databases are used for physical, chemical, and biological data with separate GIS shapefiles of sites	A true relational database is specifically designed to include data validation checks	Relational database of biological assessment data with automated data review validation tools and geospatial analysis
Biological endpoints and thresholds	Assessment based on presence or absence of targeted or key species; attainment thresholds are not specified and no BCG	A biological index or endpoint is by specific water bodies; single dimension measures used	A biological index, or model, developed and calibrated for use throughout the state for the various water body types	Biological indexes, or models, for multiple assemblages are developed and calibrated for a state and uses the BCG
Causal analysis	Support for causal analysis is lacking	Coarse indications of response via assemblage attributes at gross level (i.e., general indicator groups)	Developed indicator guilds and other aggregations to support causal associations; diagnostic capability is supported by studies	Response patterns are most fully developed and supported by extensive research and case studies across spatial and temporal scales

Development of narrative criteria

Standards are usually established narratively as well as quantitatively for particular water body types (e.g., rivers, lakes, wetlands, estuaries, etc.), sometimes specified within a geophysical region and even for specific water bodies or Ecosystem Services (ES) (e.g., Chesapeake Bay, US.; fish harvest for Canadian waters). The required number of water quality classification boundaries (i.e., categories) to be identified across a condition gradient can vary from a single pass-fail threshold to multiple acceptable classes. These multiple classes might be established to protect higher existing conditions, identify multiple impaired classes, or established to document restoration progress from severely altered to improved ecological condition. The feasibility of and need for multiple classification categories are functions of data availability and quality, the technical ability of the management programme, and public and political interests and priorities.

Development of target ecological conditions that must be quantified when setting measurable management thresholds, can be aided by narrative (non-numeric) and/or quantitative gradient models. For example, in the US, the BCG (Davies and Jackson 2006) offers, in a descriptive and standardized 6-step gradient, an ecological framework to organize and communicate detailed technical findings about changes in biological condition of water bodies across a range of human caused disturbance from undisturbed or minimally disturbed conditions to severely altered (Davies and Jackson, 2006; »Figure A 6) It provides an accounting framework for ten critical ecological attributes (»Table A 21) that show consistent patterns of change with increasing disturbance, based on measures of taxonomic composition, organism condition, spatial and temporal extent of detrimental effects, and ecosystem connectivity and function (Davies and Jackson, 2006).

Other countries or unions have developed comparable frameworks with remarkable convergence in concept and definition of incremental gradients of change such as the EU WFD ecological status classes (UKTAG 2007) and the joint Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC/ARMCANZ, 2000a).

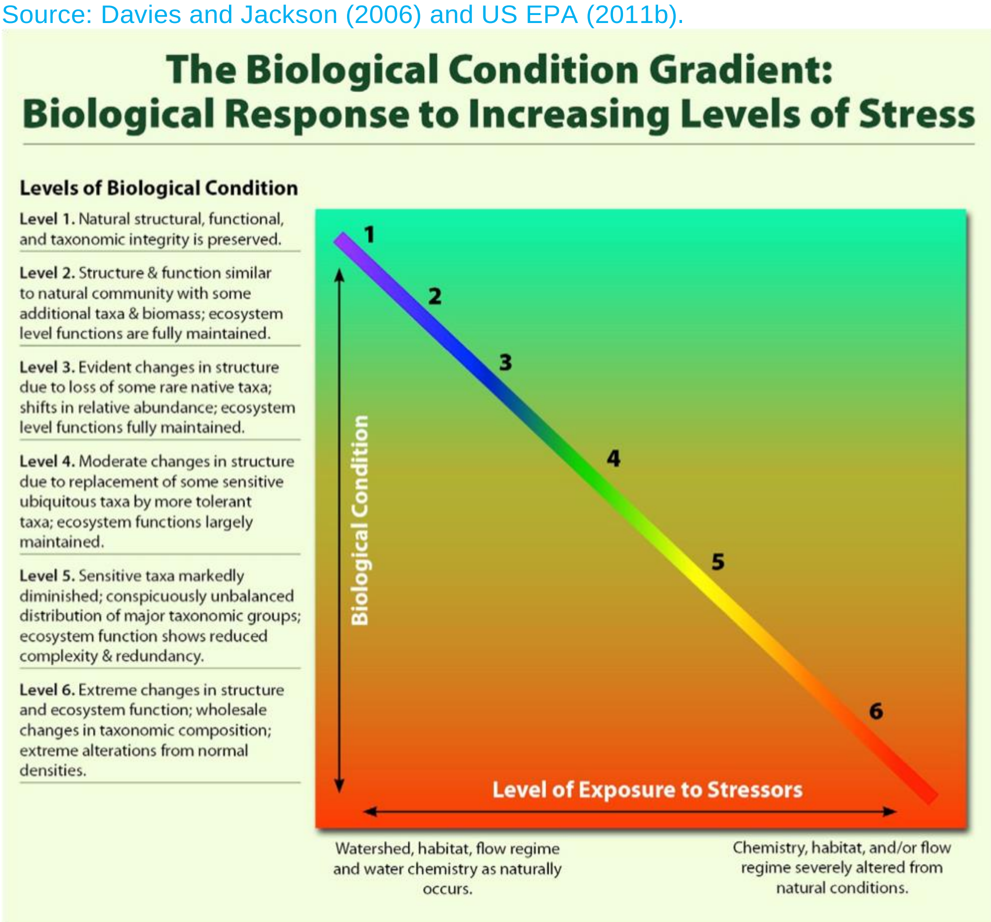
In »Chapter 3 an extensive analysis is presented concerning existing frameworks and guidelines in 15 selected countries. The methods and indicators applied for ecological assessment in the Australian/New Zealand WQGs, the EU WFDand the guidelines developed by US EPA are reviewed. In »Section 3.5 the narrative and numerical criteria applied in these jurisdictions are described.

Box A 2: Frameworks of other countries

These frameworks (see also in »Box A 2) can help to account for scientific complexity and at the same time provide a commonly understood and nationally or regionally shared vision for desired ecological condition that helps water quality managers:

- Decide what environmental conditions are desirable (goal setting)
- Interpret the environmental conditions that exist (monitoring and assessment)
- Plan for how to achieve the desired conditions and measure effectiveness of restoration (track system response to management actions)
- Communicate to stakeholders and the public (Inform management decision-making).

Figure A 6: The Biological Condition Gradient (BCG).
Source: Davies and Jackson (2006) and US EPA (2011b).



When biological response and stress information is presented in these types of ecological frameworks, it is easier for managers and the public to understand the status of the aquatic resources relative to the desired ecological goal and what is at stake when selecting management actions.

Table A 21: Biological and other ecological attributes used to characterize the Biological Condition Gradient (BCG), which is shown in »Figure A 6. The BCG method is also introduced in Section 2.7 of the preceding report. Source: Modified from Davies and Jackson (2006).

Attribute	Description
I. Historically documented, sensitive, long-lived, or regionally endemic taxa	Taxa known to have been supported according to historical, museum, or archeological records, or taxa with restricted distribution (occurring only in a location as opposed to a region), often due to unique life history requirements (e.g., sturgeon, American eel, pupfish, unionid mussel species).
II. Highly sensitive (typically uncommon) taxa	Taxa that are highly sensitive to pollution or anthropogenic disturbance. Tend to occur in low numbers, and many taxa are specialists for habitats and food type. These are the first to disappear with disturbance or pollution (e.g., most stoneflies, brook trout [in the east], brook lamprey).
III. Intermediate sensitive and common taxa	Common taxa that are ubiquitous and abundant in relatively undisturbed conditions but are sensitive to anthropogenic disturbance/pollution. They have a broader range of tolerance than attribute II taxa and can be found at reduced density and richness in moderately disturbed sites (e.g., many mayflies, many darter fish species).
IV. Taxa of intermediate tolerance	Ubiquitous and common taxa that can be found under almost any conditions, from undisturbed to highly stressed sites. They are broadly tolerant but often decline under extreme conditions (e.g., filter-feeding caddis flies, many midges, many minnow species).
V. Highly tolerant taxa	Taxa that typically are uncommon and of low abundance in undisturbed conditions but that increase in abundance in disturbed sites. Opportunistic species able to exploit resources in disturbed sites. These are the last survivors (e.g., tubificid worms, black bullhead).
VI. Nonnative or intentionally introduced species	Any species not native to the ecosystem (e.g., Asiatic clam, zebra mussel, carp, European brown trout). Additionally, there are many fish native to one part of North America that have been introduced elsewhere.
VII. Organism condition	Anomalies of the organisms; indicators of individual health (e.g., deformities, lesions, tumors).
VIII. Ecosystem finction	Processes performed by ecosystems, including primary and secondary production; respiration; nutrient cycling; decomposition; their proportion/dominance; and what components of the system carry the dominant functions. For example, shift of lakes and estuaries to phytoplankton production and microbial decomposition under disturbance and eutrophication.
IX. Spatial and temporal extent of detrimental effects	The spatial and temporal extent of cumulative adverse effects of stressors; for example, groundwater pumping in Kansas resulting in change in fish composition from fluvial dependent to sunfish
X. Ecosystem connectance	Access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation. For example, levees restrict connections between flowing water and floodplain nutrient sinks (disrupt function); dams impede fish migration, spawning.

The attributes are items that are used to characterize (help determine) the status of a system. For example, if a water body scored poorly on many of the attributes (1-10), then it would be expected to be classified as Tiers 5 or 6 on the BCG (which are Major Changes or Severe Changes).

Tiers of the Biological Condition Gradient

Tier 1: Natural or native condition.

Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.

Tier 1 represents biological conditions as they existed (or still exist) in the absence of measurable effects of stressors. The Tier 1 biological assemblages that occur in a given biogeophysical setting are the result of adaptive evolutionary processes and biogeography that selects in favor of survival of the observed species. For this reason, the expected Tier 1 assemblage of a stream from the arid southwest will be very different from that of a stream in the northern temperate forest. The maintenance of native species populations and the expected natural diversity of species are essential for Tiers 1 and 2. Non-native taxa (Attribute VI) may be present in Tier 1 if they cause no displacement of native taxa, although the practical uncertainties of this provision are acknowledged (discussed in »Section 2.2).

Attributes I and II (e.g., historically documented and sensitive taxa) can be used to help assess the status of native taxa and could be a surrogate measure to identify threatened or endangered species when classifying a site or assessing its condition.

Tier 2:Minimal changes in structure of the biotic community and minimal changes in ecosystem function.

Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

Tier 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (such as increased temperature regime or nutrient enrichment). There may be some reduction of a small fraction of highly sensitive or specialized taxa (Attribute II) or loss of some endemic or rare taxa as a result. Tier 2 can be characterized as the first change in condition from natural and it is most often manifested in nutrient enriched waters as slightly increased richness and density of sensitive ubiquitous taxa and taxa of intermediate tolerance (Attributes III and IV).

Tier 3: Evident changes in structure of the biotic community and minimal changes in ecosystem function.

Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.

Tier 3 represents readily observable changes that, for example, can occur in response to organic enrichment or increased temperature. The “evident” change in structure for Tier 3 is interpreted to be perceptible and detectable decreases in sensitive-rare or highly sensitive taxa (Attribute II) and increases in sensitive-ubiquitous taxa or opportunist organisms (Attributes III and IV). Attribute IV taxa (intermediate tolerants) may increase in abundance as an opportunistic response to nutrient inputs.

Tier 4: Moderate changes in structure of the biotic community with minimal changes in ecosystem function.

Moderate changes in structure due to replacement of some sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.

Moderate changes of structure occur as stressor effects increase in Tier 4. A substantial reduction of the two sensitive attribute groups (II and III) and replacement by more tolerant taxa (Attributes IV and V) may be observed. A key consideration is that some Attribute III sensitive taxa are maintained at a reduced level but are still an important functional part of the system (function maintained).

Tier 5: Major changes in structure of the biotic community and moderate changes in ecosystem function.

Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials.

Changes in ecosystem function (as indicated by marked changes in food-web structure and guilds) are critical in distinguishing between Tiers 4 and 5 (»Table A 22). This could include the loss of functionally important sensitive taxa and keystone taxa (Attribute I, II and III taxa) such that they are no longer important players in the system, though a few individuals may be present. Keystone taxa control species composition and trophic interactions, and are often, but not always, top predators. As an example, removal of keystone taxa by overfishing has greatly altered the structure and function even of many coastal ocean ecosystems (Jackson et al., 2001). Additionally, tolerant non-native taxa (Attribute VI) may dominate some assemblages and changes in organism condition (Attribute VII) may include significantly increased mortality, depressed fecundity, and/or increased frequency of lesions, tumors and deformities.

Tier 6: Severe changes in structure of the biotic community and major loss of ecosystem function.
Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.
Tier 6 systems are taxonomically depauperate (low diversity and/or reduced number of organisms) compared to the other tiers. For example, extremely high or low densities of organisms caused by excessive organic enrichment or severe toxicity may characterize Tier 6 systems.

Table A 22: Biological Condition Gradient: Ecological Attributes by Condition Tiers matrix. Source: Davies and Jackson (2006).

Ecological Attributes	**Biological Condition Gradient Tiers**					
	1	**2**	**3**	**4**	**5**	**6**
	Natural or native condition	**Changes in the structure of the biotic community:**				
		Minimal	**Evident**	**Moderate**	**Major**	**Severe**
		Changes in ecosystem function:				
		Minimal	**Minimal**	**Minimal**	**Moderate**	**Major**
	Native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.	Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.	Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but Sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.	Moderate changes in structure due to replacement of some Sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some Sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.	Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials.	Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.

I. Historically documented, sensitive, long-lived or regionally endemic taxa	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be absent due to global extinction or local extirpation	Some may be absent due to global, regional or local extirpation	Usually absent	Absent
II. Sensitive- rare taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Virtually all are maintained with some changes in densities	Some loss, with replaceme nt by functionally equivalent Sensitive-ubiquitous taxa	May be markedly diminished	Absent	Absent
III. Sensitive-ubiquitous taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant	Common and abundant; relative abundance greater than Sensitive-rare, taxa	Present with reproducing populations maintained; some replacement by functionally equivalent taxa of intermediate tolerance.	Frequently absent or markedly diminished	Absent
IV. Taxa of intermediate tolerance	Apredicted for natural occurrence, with at most minor changes from natural densities	As naturally present with slight increases in abundance	Often evident increases in abundanc e	Common and often abundant; relative abundance may be greater than Sensitive-ubiquitous taxa.	Often exhibit excessive dominance	May occur in extremely high OR extremely low densities; richness of all taxa is low
V. Tolerant taxa	As naturally occur, with at most minor changes from natural densities	As naturally present with slight increases in abundance	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often extreme departures from normal densities (high or low).

Annex 4

Normative definitions of ecological status classification in the European Water Framework Directive (EU WFD)

Table A 23: Ecological status classification used in the EU WFD.
Source: EC (2000a).

Ecological status	Element	High status	Good status	Moderate status
In rivers, lakes, transitional waters and coastal waters	General definition	The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion. These are the type-specific conditions and communities.	The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions.	The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions.
Biological quality elements in rivers	Phytoplankton	The taxonomic composition of phytoplankton corresponds totally or nearly totally to undisturbed conditions. The average phytoplankton abundance is wholly consistent with the type-specific physico-chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type- specific physicochemical conditions.	There are slight changes in the composition and abundance of planktonic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbances to the balance of organisms present in the water body or to the physico- chemical quality of the water or sediment. A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	The composition of planktonic taxa differs moderately from the type-specific communities. Abundance is moderately disturbed and may be such as to produce a significant undesirable disturbance in the values of other biological and physico-chemical quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
	Macrophytes and Phytobenthos	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in the average macrophytic and the average phytobenthic abundance.	There are slight changes in the composition and abundance of macrophytic and phytobenthic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbances to the balance of organisms present in the water body or to the physico-chemical quality of the water or sediment. The phytobenthic community is not adversely affected by bacterial tufts and coats present due to anthropogenic activity.	The composition of macrophytic and phytobenthic taxa differs moderately from the type-specific community and is significantly more distorted than at good status. Moderate changes in the average macrophytic and the average phytobenthic abundance are evident. The phytobenthic community may be interfered with and,in some areas, displaced by bacterial tufts and coats present as a result of anthropogenic activities.
	Benthic invertebr	The taxonomic composition and abundance correspond totally or	There are slight changes in the composition and abundance of	The composition and abundance of invertebrate taxa differ

	ate fauna	nearly totally to undisturbed conditions. The ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.	invertebrate taxa from the type-specific communities. The ratio of disturbance-sensitive taxa to insensitive taxa shows slight alteration from type-specific levels. The level of diversity of invertebrate taxa shows slightsigns of alteration from type- specific levels.	moderately from the type-specific communities. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance-sensitive taxa to insensitive taxa, and the level of diversity, are substantially lower than the type-specific level and significantly lower than for good status.
	Fish fauna	Species composition and abundance correspond totally or nearly totally to undisturbed conditions. All the type-specific disturbance-sensitive species are present. The age structures of the fish communities show little sign of anthropogenic disturbance and are not indicative of a failure in the reproduction or development of any particular species.	There are slight changes in species composition and abundance from the type-specific communities attributable to anthropogenic impacts on physicochemical and hydromorphological quality elements. The age structures of the fish communities show signs of disturbance attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements, and, in a few instances, are indicative of a failure in the reproduction or development of a particular species, to the extent that some age classes may be missing.	The composition and abundance differ moderately from the type-specific communities attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements. The age structure of the fish communities shows major signs of anthropogenic disturbance, to the extent that a moderate proportion of the type specific species are absent or of very low abundance.

Ecological status	Element	High status	Good status	Moderate status
Hydro-morpho-logical quality elements in rivers	Hydrological regime	The quantity and dynamics of flow, and the resultant connection to groundwaters, reflect totally, or nearly totally, undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	River continuity	The continuity of the river is not disturbed by anthropogenic activities and allows undisturbed migration of aquatic organisms and sediment transport.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Morphological conditions	Channel patterns, width and depth variations, flow velocities, substrate conditions and both the structure and condition of the riparian zones correspond totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Physico-chemical quality elemements in rivers	General conditions	The values of the physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Levels of salinity, pH, oxygen balance, acid neutralisingcapacity and temperature do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions.	Temperature, oxygen balance, pH, acid neutralizing capacity and salinity do not reach levels outside therange established so as to ensure the functioning of the type specific ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Specific synthet	Concentrations close to zero and at least below the limits of	Concentrations not in excess of the standards set in accordance	Conditions consistent with the achievement of the values

	ic pollutants	detection of the most advanced analytical techniques in general use.	with the procedure detailed in section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	specified above for the biological quality elements.
	Specific non-synthetic pollutants	Concentrations remain within the range normally associated with undisturbed conditions (background levels = bgl).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 (2) without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Ecological status	**Element**	**High status**	**Good status**	**Moderate status**
Biological quality elements in lakes	Phytoplankton	The taxonomic composition and abundance of phytoplankton correspond totally or nearly totally to undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico- chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type specific physicochemical conditions.	There are slight changes in the composition and abundance of planktonic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico- chemical quality of the water or sediment. A slight increase in the frequency and intensity of the type specific planktonic blooms may occur.	The composition and abundance of planktonic taxa differ moderately from the type-specific communities. Biomass is moderately disturbed and may be such as to produce a significant undesirable disturbance in the condition of other biological quality elements and the physico- chemical quality of the water or sediment. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
	Macrophytes and Phytobenthos	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in the average macrophytic and the average phytobenthic abundance.	There are slight changes in the composition and abundance of macrophytic and phytobenthic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water. The phytobenthic community is not adversely affected by bacterial tufts and coats present due to anthropogenic activity.	The composition of macrophytic and phytobenthic taxa differ moderately from the type-specific communities and are significantly more distorted than those observed at good quality. Moderate changes in the average macrophytic and the average phytobenthic abundance are evident. The phytobenthic community may be interfered with, and, in some areas, displaced by bacterial tufts and coats present as a result of anthropogenic activities.
	Benthic invertebrate fauna	The taxonomic composition and abundance correspond totally or nearly totally to the undisturbed conditions. The ratio of disturbance sensitive taxa to insensitive taxa shows no signs of alteration from undisturbed levels. The level of diversity of invertebrate taxa shows no sign of alteration from undisturbed levels.	There are slight changes in the composition and abundance of invertebrate taxa compared to the type-specific communities. The ratio of disturbance sensitive taxa to insensitive taxa shows slight signs of alteration from type-specific levels. The level of diversity of invertebrate taxa shows slight signs of alteration from type-specific levels.	The composition and abundance of invertebrate taxa differ moderately from the type-specific conditions. Major taxonomic groups of the type-specific community are absent. The ratio of disturbance sensitive to insensitive taxa, and the level of diversity, are substantially lower than thetype-specific level and significantly lower than for good status.
	Fish fauna	Species composition and abundance correspond totally or nearly totally to undisturbed conditions. All the type-specific sensitive species are present. The age structures of the fish communities show little sign of anthropogenic disturbance and are not indicative of a failure in the reproduction or development of	There are slight changes in species composition and abundance from the type-specific communities attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements. The age structures of the fish communities show signs of disturbance attributable to anthropogenic impacts on physico-chemical	The composition and abundance of fish species differ moderately from the type-specific communities attributable to anthropogenic impacts on physico-chemical or hydromorphological quality elements. The age structure of the fish communities shows major signs of disturbance, attributable to anthropogenic impacts on physico-chemical or

		a particular species.	or hydromorphological quality elements, and, in a few instances, are indicative of a failure in the reproduction or development of a particular species, to the extent that some age classes may be missing.	hydromorphological quality elements, to the extent that a moderate proportion of the type specific species are absent or of very low abundance.
Hydrom-orpholog ical quality elements in lakes	Hydrologic al regime	The quantity and dynamics of flow, level, residence time, and the resultant connection to groundwaters, reflect totally or nearly totally undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Morpholo gical conditions	Lake depth variation, quantity and structure of the substrate, and both the structure and condition of the lake shore zone correspond totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Ecologic al status	**Element**	**High status**	**Good status**	**Moderate status**
Physico-chemical quality element s in lakes	General conditions	The values of physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Levels of salinity, pH, oxygen balance, acid neutralizing capacity, transparency and temperature do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions.	Temperature, oxygen balance, pH, acid neutralising capacity, transparency and salinity do not reach levels outside the range established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Specific synthetic pollutants	Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Specific non-synthetic pollutants	Concentrations remain within the range normally associated with undisturbed conditions (background levels = bgl).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 (2) without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Biological quality elements in transitional waters	Phytoplankt on	The composition and abundance of the phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico- chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type specific physicochemical conditions.	There are slight changes in the composition and abundance of phytoplanktonic taxa. There are slight changes in biomass compared to the type-specific conditions. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico- chemical quality of the water. A slight increase in the frequency and intensity of thetype specific planktonic blooms may occur.	The composition and abundance of phytoplanktonic taxa differ moderately from type-specific conditions. Biomass is moderately disturbed and may be such as to produce a significant undesirable disturbance in the condition of other biological quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
	Macroalgae	The composition of macroalgal taxa is consistent	There are slight changes in the composition and abundance of	The composition of macroalgal taxa differs moderately from

		with undisturbed conditions. There are no detectable changes in macroalgal cover due to anthropogenic activities.	macroalgal taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water.	type-specific conditions and is significantly more distorted than at good quality. Moderate changes in the average macroalgal abundance are evident and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.
	Angiosperms	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in angiosperm abundance due to anthropogenic activities.	There are slight changes in the composition of angiosperm taxa compared to the type- specific communities. Angiosperm abundance shows slight signs of disturbance.	The composition of the angiosperm taxa differs moderately from the type-specific communities and is significantly more distorted than at good quality. There are moderate distortions in the abundance of angiosperm taxa.
	Benthic invertebrate fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.	The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions. Most of the sensitive taxa of the type-specific communities are present.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the type-specific conditions. Taxa indicative of pollution are present. Many of the sensitive taxa of the type-specific communities are absent.
	Fish fauna	Species composition and abundance is consistent with undisturbed conditions.	The abundance of the disturbance-sensitive species shows slight signs of distortion from type-specific conditions attributable to anthropogenic impacts on physicochemical or hydromorphological quality elements.	A moderate proportion of the type-specific disturbance-sensitive species are absent as a result of anthropogenic impacts on physicochemical or hydromorphological quality elements.

Ecological status	**Element**	**High status**	**Good status**	**Moderate status**
Hydro-morpho-logical quality elements in transitional waters	Tidal regime	The freshwater flow regime corresponds totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Morphological conditions	Depth variations, substrate conditions, and both the structure and condition of the intertidal zones correspond totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Physico-chemical quality elements in transitional waters	General conditions	Physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally associated with undisturbed conditions. Temperature, oxygen balance and transparency do not show signs of anthropogenic disturbance and remain within the range normally associated with undisturbed conditions.	Temperature, oxygenation conditions and transparency do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Specific synthetic pollutants	Concentrations close to zero and at least below the limits of detection of the most	Concentrations not in excess of the standards set in accordance with the procedure detailed in	Conditions consistent with the achievement of the values specified above for the

		advanced analytical techniques in general use.	section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	biological quality elements.
	Specific non-synthetic pollutants	Concentrations remain within the range normally associated with undisturbed conditions (background levels = bgl).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 (2) without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Ecological status	**Element**	**High status**	**Good status**	**Moderate status**
Biological quality elements in coastal water	Phytoplankton	The composition and abundance of phytoplanktonic taxa are consistent with undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physico- chemical conditions and is not such as to significantly alter the type-specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type specific physicochemical conditions.	The composition and abundance of phytoplanktonic taxa show slight signs of disturbance. There are slight changes in biomass compared to type-specific conditions. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the quality of the water. A slight increase in the frequency and intensity of the type-specific planktonic blooms may occur.	The composition and abundance of planktonic taxa show signs of moderate disturbance. Algal biomass is substantially outside the range associated with type-specific conditions, and is such as to impact upon other biological quality elements. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
	Macroalgae and angiosperms	All disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present. The levels of macroalgal cover and angiosperm abundance are consistent with undisturbed conditions.	Most disturbance-sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are present. The level of macroalgal cover and angiosperm abundance show slight signs of disturbance.	A moderate number of the disturbance- sensitive macroalgal and angiosperm taxa associated with undisturbed conditions are absent. Macroalgal cover and angiosperm abundance is moderately disturbed and may be such as to result in an undesirable disturbance to the balance of organisms present in the water body.
	Benthic invertebrate fauna	The level of diversity and abundance of invertebrate taxa is within the range normally associated with undisturbed conditions. All the disturbance-sensitive taxa associated with undisturbed conditions are present.	The level of diversity and abundance of invertebrate taxa is slightly outside the range associated with the type-specific conditions. Most of the sensitive taxa of the type-specific communities are present.	The level of diversity and abundance of invertebrate taxa is moderately outside the range associated with the type-specific conditions. Taxa indicative of pollution are present.Many of the sensitive taxa of the type-specific communities are absent.
Hydro-morpho-logical quality elements in coastal water	Tidal regime	The freshwater flow regime and the direction and speed of dominant currents correspond totally or nearly totally to undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Morphological Conditions	The depth variation, structure and substrate of the coastal bed, and both the structure and condition of the inter-tidal zones correspond totally or nearly totally to the undisturbed conditions.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Physico-chemical quality elements in coastal water	General conditions	The physico-chemical elements correspond totally or nearly totally to undisturbed conditions. Nutrient concentrations remain within the range normally	Temperature, oxygenation conditions and transparency do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem and the	Conditions consistent with the achievement of the values specified above for the biological quality elements.

		associated with undisturbed conditions. Temperature, oxygen balance and transparency do not show signs of anthropogenic disturbance and remain within the ranges normally associated with undisturbed conditions.	achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	
	Specific synthet ic polluta nts	Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.
	Specific non-synthetic pollutants	Concentrations remain within the range normally associated with undisturbed conditions (background levels = bgl).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 (2) without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Ecologic al status	Element	High status	Good status	Moderate status
Biological quality elements, ecological potential for heavily modified or artificial water bodies	------------------	The values of the relevant biological quality elements reflect, as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the waterbody.	There are slight changes in the values of the relevant biological quality elements as compared to the values found at maximum ecological potential.	There are moderate changes in the values of the relevant biological quality elements as compared to the values found at maximum ecological potential. These values are significantly more distorted than those found under good quality.
Hydro-morpho-logical elements, ecological potential for heavily modified or artificial water bodies	------------------	The hydromorphological conditions are consistent with the only impacts on the surface water body being those resulting from the artificial or heavily modified characteristics of the water body once all mitigation measures have been taken to ensure the best approximation to ecological continuum, in particular with respect to migration of fauna and appropriate spawning and breeding grounds.	Conditions consistent with the achievement of the values specified above for the biological quality elements.	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Physico-chemical elements, ecological potential for heavily modified or artificial water bodies	------------------	------------------	------------------	------------------
General conditions, ecological potential for heavily modified or artificial water bodies	------------------	Physico-chemical elements correspond totally or nearly totally to the undisturbed conditions associated with the surface water body type most closely comparable to the artificial or heavily modified body concerned. Nutrient concentrations remain within the range normally associated with such	The values for physico-chemical elements are within the ranges established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements. Temperature and pH do not reach levels outside the ranges established so as to ensure the functioning of the ecosystem	Conditions consistent with the achievement of the values specified above for the biological quality elements.

		undisturbed conditions. The levels of temperature, oxygen balance and pH are consistent with the those found in the most closely comparable surface water body types under undisturbed conditions.	and the achievement of the values specified above for the biological quality elements. Nutrient concentrations do not exceed the levels established so as to ensure the functioning of the ecosystem and the achievement of the values specified above for the biological quality elements.	
Specific synthetic pollutants, ecological potential for heavily modified or artificial water bodies	------------------	Concentrations close to zero and at least below the limits of detection of the most advanced analytical techniques in general use.	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 without prejudice to Directive 91/414/EC and Directive 98/8/EC. (<EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.
Specific non-synthetic pollutants, ecological potential for heavily modified or artificial water bodies	------------------	Concentrations remain within the range normally associated with the undisturbed conditions found in the surface water body type most closely comparable to the artificial or heavily modified body concerned (background levels = bgl).	Concentrations not in excess of the standards set in accordance with the procedure detailed in section 1.2.6 (3) without prejudice to Directive 91/414/EC and Directive 98/8/EC. (< EQS)	Conditions consistent with the achievement of the values specified above for the biological quality elements.

Ecological status	Element	High status	Good status	Moderate status
Groundwater quantitative status, Parameters for classi-fication: Groundwater level regime	Groundwater level		The level of groundwater in the groundwater body is such that the available groundwater resource is not exceeded by the long-term annual average rate of abstraction. Accordingly, the level of groundwater is not subject to anthropogenic alterations such as would result in: • failure to achieve the environmental objectives specified under Article 4 for associated surface waters, • any significant diminution in the status of such waters, • any significant damage to terrestrial ecosystems which depend directly on the groundwater body, and alterations to flow direction resulting from level changes may occur temporarily, or continuously in a spatially limited area, but such reversals do not cause saltwater or other intrusion, and do not indicate a sustained and clearly identified anthropogenically induced trend in flow direction likely toresult in such intrusions.	
Groundwater	General		The chemical composition of the	

chemical status, parameters for deter-mination: conductivity, concen-trations of pollutants			groundwater body is such that the concentrations of pollutants: • as specified below, do not exhibit the effects of saline or other intrusions • do not exceed the quality standards applicable under other relevant Community legislation in accordance with Article 17 • are not such as would result in failure to achieve the environmental objectives specified under Article 4 for associated surface waters nor any significant diminution of the ecological or chemical quality of such bodies nor in any significant damage to terrestrial ecosystems which depend directly on the groundwater body	
	Conductivity		Changes in conductivity are not indicative of saline or other intrusion into the groundwater body	

[1] The following abbreviations are used: bgl = background level, EQS = environmental quality standard.

[2] Application of the standards derived under this protocol shall not require reduction of pollutant concentrations below background levels: (EQS >bgl).

[3] Application of the standards derived under this protocol shall not require reduction of pollutant concentrations below background levels

Table A 24: Indicators used in the EU WFD.
Source: EC (2000a).
This table shows indicators for the classifications for surface water. For groundwater, parameter for the classification of quantitative status is just groundwater level regime.

Quality elements for the classification of ecological status	Ecological Status	Rivers	Lakes	Transitional waters	Coastal waters	Artificial and heavily modified surface water bodies
Biological elements		• Composition and abundance of aquatic flora • Composition and abundance of benthic invertebrate fauna • Composition, abundance and age structure of fish fauna	• Composition, abundance and biomass of phytoplankton • Composition and abundance of other aquatic flora • Composition and abundance of benthic invertebrate fauna • Composition, abundance and age structure of fish fauna	• Composition, abundance and biomass of phytoplankton • Composition and abundance of other aquatic flora • Composition and abundance of benthic invertebrate fauna • Composition and abundance of fish fauna	• Composition, abundance and biomass of phytoplankton • Composition and abundance of other aquatic flora • Composition and abundance of benthic invertebrate fauna	The quality elements applicable to artificial and heavily modified surface water bodies shall be those applicable to whichever of the four natural surface water categories here most

Hydro-morphologic alelements supporting the biological elements	Hydrological regime in rivers and lakes	• quantity & dynamics of water flow	• quantity and dynamics of water flow	• freshwater flow • wave exposure	• direction of dominant currents • wave exposure	closely resembles the heavily modified or artificial water body concerned.
	Trial regime in other water bodies	• connection to groundwater bodies	• residence time • connection to the groundwater body		• wave exposure	
	River continuity					
	Morphological conditions	• river depth and width variation • structure and substrate of the river bed • structure of the riparian zone	• lake depth variation • quantity, structure and substrate of the lake bed • structure of the lake shore	• depth variation • quantity, structure and substrate of the bed • structure of the intertidal zone	• depth variation • structure and substrate of the coastal bed • structure of the intertidal zone	
Chemical and physico-chemical elements supporting the biological elements	General	• Thermal conditions • Oxygenation conditions • Salinity • Acidification status • Nutrient conditions	• Transparency • Thermal conditions • Oxygenation conditions • Salinity • Acidification status • Nutrient conditions	• Transparency • Thermal conditions • Oxygenation conditions • Salinity • Nutrient conditions	• Transparency • Thermal conditions • Oxygenation conditions • Salinity • Nutrient conditions	
Chemical and physico-chemical elements supporting the biological elements	Specific pollutants	• Pollution by all priority substances identified as being discharged into the body of water • Pollution by other substances identified as being discharged in significant quantities into the body of water	• Pollution by all priority substances identified as being discharged into the body of water • Pollution by other substances identified as being discharged in significant quantities into the body of water	• Pollution by all priority substances identified as being discharged into the body of water • Pollution by other substances identified as being discharged in significant quantities into the body of water	• Pollution by all priority substances identified as being discharged into the body of water • Pollution by other substances identified as being discharged in significant quantities into the body of water	

Table A 25: Classifications of water bodies typology adopted in the EU WFD.
Source: EC (2000a).
* System A: In this system, the surface water bodies within the river basin district shall first be differentiated by the relevant ecoregions in accordance with the geographical characters of surface body type, such as altitude, size based on catchment area and geology.

** System B: The same degree of differentiation is used as in system A, but it considers additionally the values for the obligatory descriptors and such optional descriptors, or combinations of descriptors, as are required to ensure that type specific biological reference conditions (RCs) can be reliably derived.

Surface Water				
Ecoregions & surface water body types Fixed typology	**Rivers**	**Lakes**	**Transitional Waters**	**Coastal waters**
Ecoregion (System A*)	Ecoregions shown on map A in Annex XI (page 71)	Ecoregions shown on map A in Annex XI (page 71)	The following as identified on map B in Annex XI (page 72): Baltic Sea Barents Sea, Norwegian Sea North Sea North Atlantic Ocean Mediterranean Sea	The following as identified on map B in Annex XI (page 72): Baltic Sea Barents Sea Norwegian Sea North Sea North Atlantic Ocean Mediterranean Sea
Type (System A)	**Altitude typology (m)** high: >800 mid-altitude: 200–800 lowland: <200 **Size typology based on catchment area (km^2)** small: 10 to 100 medium: >100 to 1000 large: >1000 to 10000 very large: >10000 **Geology** Calcareous, siliceous, organic	**Altitude typology (m)** high: >800 mid-altitude: 200–800 lowland: <200 m **Depth typology based on mean depth (m)** <3, 3–15, >15 m Size typology based on catchment area (km2) 0,5–1, 1–10, 10–100, >100 **Geology** Calcareous, siliceous, organic	**Based on mean annual salinity** <0,5.: freshwater 0,5 to <5.: oligohaline 5 to <18.: mesohaline 18 to <30.: polyhaline 30 to <40.: euhaline **Based on mean tidal range (m)** <2 m: microtidal 2–4 m: mesotidal >4 m: macrotidal	**Based on mean annual salinity** <0,5.: freshwater 0,5 to <5.: oligohaline 5 to <18.: mesohaline 18 to <30.: polyhaline 30 to <40.: euhaline **Based on mean depth (m)** shallow waters: <30 m intermediate: 30–200 m deep: >200 m
Alternative characterization (System B**)	Physical and chemical factors that determine the characteristics of the river or part of the river and hence the biological population structure and composition	Physical and chemical factors that determine the characteristics of the lake and hence the biological population structure and composition	Physical and chemical factors that determine the characteristics of the transitional water and hence the biological population structure and composition	Physical and chemical factors that determine the characteristics of the coastal water and hence the biological community structure and composition
Obligatory factors (System B)	• altitude • latitude • longitude • geology • size	• altitude • latitude • longitude • depth • geology • size	• latitude • longitude • tidal range • salinity	• Latitude • Longitude • tidal range • salinity

Surface Water				
Ecoregions & surface water body types Fixed typology	**Rivers**	**Lakes**	**Transitional Waters**	**Coastal waters**
Optional factors (System B)	• distance from river source • energy of flow (function of flow and slope) • mean water width • mean water depth • mean water slope • form and shape of main river bed • river discharge (flow) category • valley shape • transport of solids • acid neutralizing capacity • mean substratum	• mean water depth • lake shape • residence time • mean air temperature • air temperature range • mixing characteristics (e.g. monomictic, dimictic, polymictic) • acid neutralising capacity • background nutrient status • mean substratum composition • water level fluctuation	• depth • current velocity • wave exposure • residence time • mean water temperature • mixing characteristics • turbidity • mean substratum composition • shape • water temperature range	• current velocity • wave exposure • mean water temperature • mixing characteristics • turbidity • retention time (of enclosed bays) • mean substratum composition • water temperature range

Typology	Ground Waters
Initial characterisation	• the location and boundaries of the groundwater body or bodies • the pressures to which the groundwater body or bodies are liable to be subject including: • diffuse sources of pollution • point sources of pollution • abstraction • artificial recharge • the general character of the overlying strata in the catchment area from which the groundwater body receives its recharge • those groundwater bodies for which there are directly dependent surface water ecosystems or terrestrial ecosystems.
Further characterisation	• geological characteristics of the groundwater body including the extent and type of geological units, • hydrogeological characteristics of the groundwater body including hydraulic conductivity, porosity and confinement • characteristics of the superficial deposits and soils in the catchment from which the groundwater body receives its recharge, including the thickness, porosity, hydraulic conductivity, and absorptive properties of the deposits and soils • stratification characteristics of the groundwater within the groundwater body, • an inventory of associated surface systems, including terrestrial ecosystems and bodies of surface water, with which the groundwater body is dynamically linked • estimates of the directions and rates of exchange of water between the groundwater body and associated surface systems • sufficient data to calculate the long term annual average rate of overall recharge, • characterisation of the chemical composition of the groundwater, including specification of the contributions from human activity. Member States may use typologies for groundwater characterization when establishing natural background levels for these bodies of groundwater.

Typology	Ground Waters
Review of the impact of human activity on groundwaters	• the location of points in the groundwater body used for the abstraction of water with the exception of: • points for the abstraction of water providing less than an average of 10 m3 per day, or • points for the abstraction of water intended for human consumption providing less than an average of 10 m3 per day or serving less than 50 persons • the annual average rates of abstraction from such points • the chemical composition of water abstracted from the groundwater body, • the location of points in the groundwater body into which water is directly discharged, • the rates of discharge at such points • the chemical composition of discharges to the groundwater body, and
Review of the impact of changes in groundwater levels	• surface water and associated terrestrial ecosystems • water regulation, flood protection and land drainage • human development
Review of the impact of pollution on groundwater quality	• the body of groundwater is so polluted that achieving good groundwater chemical status is infeasible or disproportionately expensive.

Comparing these two systems, System B has the advantage of considering more information to ensure reliable derivation for specific biological RCs, and allowed for more flexibility in adopting ecologically meaningful typologies. See Moss et al. (2003) for further discussion and rationale for applying a more realistic approach to typology than the more prescriptive System A. While the principle of using a water body typology has considerable theoretical merit in providing for a classification system that allows for comparison of water bodies that are similar in nature and against a RC benchmark, the identification of biological communities that represent type-specific RC is, inevitably, an approximation of reality. For that reason classification has been recommended by some workers to be based on RCs within a water body (Carvalho et al., 2009; Jyväsjärvi J et al., 2009) rather than the wider type-specific approach that would follow the original logic of the EU WFD.

Annex 5

Utilizing the Ecosystem Services Approach for Water Framework Directive Implementation (ESAWADI): project summary

The potential of Ecosystem Services (ES) has been tested for the application of defining and monitoring of a target state in the context of the European Water Framework Directive (EU WFD) by the Ecosystem Services Approach for the Water Framework Directive (ESAWADI) project (ESAWADI, 2013). The EU WFD's major goal is the achievement of a good state of all European waters. The project team promoted an Ecosystem Service Approach (ESA) which was neither merely anthropocentric (focused on human benefits maximisation) nor ecosystem-centred (conservation without taking human needs into consideration). Rather was it oriented towards a sustainable co-evolution between nature and society (using a combination of scientific expertise and stakeholder consultations). The EU WFD uses natural reference conditions (RCs) as target states but allows exemptions for (heavily) modified water bodies where improvements cannot be achieved at acceptable costs. A review of conceptual foundations, insights on potential and limitations, results from case studies can be found in the ESAWADI reports (ESAWADI 2011, 2013).

In the following, major findings of the ESAWADI project for using the ESA for policy implementation are summarized. The experience of the ESAWADI team was that the principal strengths of the ESA lie in its structured and systematic approach to describing the way functioning ecosystems provide benefits to society. ESA may ensure that a comprehensive and consistent approach is used to highlight the linkages between uses and ecosystem functions, thereby identifying the full range of ES (potential or existing services), and thus facilitating the design of relevant policies. However, assessing all ES in a watershed presents a huge operational challenge: water managers expressed concern that this was too complex and required too much work. This may present an essential drawback in the eyes of those who are of the opinion that ESA is not useful if it does not produce quantitative or monetary results. Still, the ESAWADI project has demonstrated in all case studies that a thorough quantification and valuation of ES, aiming at "full monetarization“, is neither feasible nor desirable.

The potential role of ESA as a support for communication and environmental education was clearly demonstrated. Water managers and other stakeholders involved in the project appreciated that ESA is a good educational and participatory tool, helping to create common ground with respect to the potential of a healthy ecosystem, benefits of ecosystems protection and restoration, awareness raising and discussions on ecological processes and the potential services. The case studies have proven that ESA can be a valuable planning tool for systematically identifying all stakeholders and possible conflicts, as well as for illustrating the diverse benefits a measure could generate. In particular, water managers considered the representation of cultural ES as a true added value of the ESA.

Table A 26: Examples of indicators relevant to aquatic ecosystems.
Source: ESAWADI (2013).

	Potential Services	Ecosystems	Ideal Indicators
Regulating services	Reducing the vulnerability of goods and people in relation to drought	Bank	• *Water retention time before salting out: depending on the nature and the soil structure* • *Evolution of the low water functions of the state of the considered compartment*
		Low water bed	
		Oxbow plains, wetlands…	
		Alluvial groundwater	• *Piezometric assessment*
	Water quality control	Bank	• *Soil hydromorphology* • *Organic matter and micro pollutants storage and retention capacity* • *Microbial biomass activity* • *Oxygen concentration in the water* • *Diversity and abundance of micro organisms and aquatic invertebrates* • *Surface of riffles zone, sand…* • *Fishes habitat quality* • *Number of authorizations delivered, quantity of commercial fish species*
		Low water bed	
		Oxbow plains, wetlands …	
		Alluvial groundwater	
Supply services	Professional fishing	Bank	
		Low water bed	
		Oxbow plains, wetlands …	
		Alluvial groundwater	
	Agricultural production	Bank	• *Land cover: type of crops, crop area…* • *Yields*
		Low water bed	
		Oxbow plains, wetlands…	
		Alluvial groundwater	
Social-type services	Water sports	Bank	• *Number of people practicing water activities like canoe…* • *Spillover from these activities*
		Low water bed	
		Oxbow plains, wetlands…	
		Alluvial groundwater	

Table A 27: Indicators for quantifying selected Ecosystem Services (ES) on the middle stretch of the Dordogne.
Source: ESAWADI (2013)

Ecosystem service and hydroelectricity		**Indicators**
Water resource availability	Benefit derived from ES by humans	• Volume of water drawn for irrigation (low water level and/or annual) • Number of farmers/farms involved • Area of irrigated land (quantity) • Types of crops on irrigated land (diversity)
	Effective ES	• Amount of water available in the river that can be drawn without damaging its good status. Data based on different

	Costs of preserving resources supported by the users	• Charge for domestic use
Provision of fertile soil	Benefit derived from ES by humans	• Turnover of agricultural crops on the banks of the river • Quantity, quality and diversity of harvests • Production quality (labels, controlled appellations, etc.)
	Effective ES	• Number of hectares on the banks of the river
Provision of fish	Benefit derived from ES by humans	• Annual catch from professonal fishing • Number of professional fisherman • Socio-economic profiles of professional fisherman • Annual turnover of professional fishing
	Potential ES	• Presence and characteristics of oxbow lakes • Number of potential spawning sites • Number of fish stradings
	Costs of preserving resources	• Charge of fisherman
Creation of a specific landscape	Benefit derived from ES by humans	• Contribution of the landscape quality in the construction of a territorial identity • Landscape quality (wild/natural appearance)- perceptions
	Effective ES	• Presence and abundance of invasive species (Ash-leaf maple)- loss of endemic species and homogenization of the
	Potential ES	• Presence of oxbow lakes, animal and plant biodiversity

Ecosystem service and hydroelectricity		**Indicators**
Creation of conditions favorable to leisure activities	Benefit derived from ES by humans	• Use of canoeing sites • Numbers of participants in clubs • Number of clubs • Presence of a race course • Number of bathing areas developed • Popularity and beaches • Number of camp sites with beaches
	Effective ES	• Number of recovery points • Water level (flow)
Maintenance of water quality	Benefit derived from ES by humans	• Number of days when bathing, canoeing and fishing is banned • Water quality at the withdrawal points • Strength of the water treatment for drinking water supply • Price of water
	Effective ES	• Self-purifying capacity

	Potential ES	• Presence and number of riffles

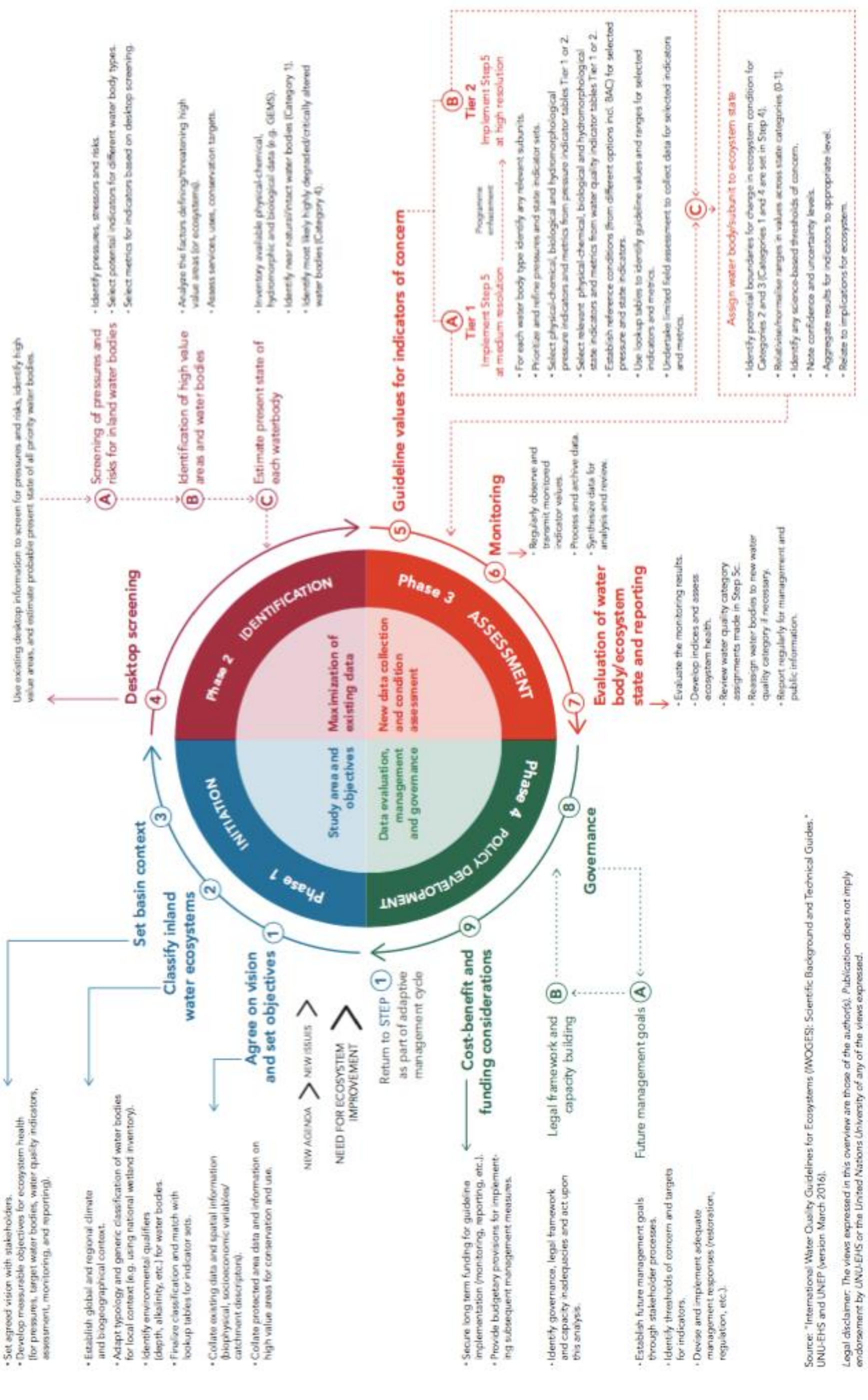
Table A 28 Ecosystem services linked to ecological processes, natural resources, natural attributes and habitats.
Source: ESAWADI (2013).

Regulation Processes and Services (direct)				Functional Compartments									Habitats					
Processes				Lower water bed	riffle	Pool bar	alluvium	River banks	Floodplain	Oxbows/Wetlands	Alluvial groundwater	Hyporheic zone	Aquatic Vegetation	Alluvial forest covers	Timber debris	Stream side Vegetation	Flood prone meadows	Effectiveness of the Dordogne Basin
Conservation of water resource quality – Conservation of water and aquatic habitat quality	Self-purification process	Dilution of pollutants (volume)		•														✓
		Suspended matters deposit	Currents slowdown			•												✓
			Water mechanical filtration in the gravel interstices		•		•					•	•			•		✓
		Water oxygenation	Water mechanical mixing		•								•			•		✓
			Photosynthesis (diurnal activity)															✓
		Meanders' formation → Creation of a turbulence area extrados of meanders	Favour water mixing and gaseous exchange between hydrosphere and atmosphere	•														✓
		Biochemical degradation (bacteria, UV rays, etc.) of organic matter and of certain micropollutants that can be metabolized		•	•							•						✓
		Micropollutants capture (in mineralized organic matters) and bioaccumulation by transformed or not living organisms		•								•						✓
		Absorption and transformation by living organism of organic matters and of certain pollutants that are under different forms		•								•	•			•		✓
	Buffer zone	Capture, transformation and degradation (form of state change) of organic matter and pollutants in seepage water	Biochemical degradation in soils, assimilation by living organisms, sedimentation						•	•								✓
		Mechanical filtration of seepage water	Slower flow speeds in the ground					•	•	•								✓
		Absorption of nutritive elements by vegetation (runoff and seepage water)												•		•	•	✓
		Mechanical filtration of runoff water	Capture of organic and mineral particles							•				•		•	•	✓
		Thermic regulation of surface water	Avoid important increases in temperatures (which has an influence on the oxygen rate) and aquatic plants development control → limitation of eutrophication problems									•		•				✓
		Shade of the lower water bed												•				✓

Regulation Processes and Services (direct)			Functional Compartments									Habitats					
Processes			Lower water bed	riffle	Pool bar	alluvium	River banks	Floodplain	Oxbows/Wetlands	Alluvial groundwater	Hyporheic zone	Aquatic Vegetation	Alluvial forest covers	Timber debris	Stream side Vegetation	Flood prone meadows	Effectiveness of the Dordogne Basin
Conservation of soil quality	Flooding	River overflowing → silt deposit	•														±
	Mobilisation of rivers external sediments	Avoid the incision of the water bed → lowering of the groundwater level	•														✓
Conservation of alluvial lands and terraces	Mobilisation of rivers external sediments	Avoid river banks erosion	•														✓
	Energy dissipation because of roughness, slowing of hydraulic sediments	Dissipation of erosive flows					•		•								✓
		Stabilisation of river bed and river banks bases										•			•		
	Capture of organic and mineral particles carried by runoff water											•			•	•	✓
Stability of infrastructures (bridges, roads, etc.)	Mobilisation of rivers external sediments	Avoid river banks erosion	•														✓
Reduction of flood impacts	Flood expansion area																±
	Flow slow down during flooding events	Hydraulic flows energy dissipation by increasing roughness															✓
		Flood capping: flooding peak mitigation															✓
		Flooding peaks desynchronization															✓
Availability of water resource- Conservation of water levels	Groundwater recharge		•						•		•						✓
	Atmospheric water catching and return to the river	Seepage influenced by runoff water slow down due to present vegetation						•	•								✓
		Runoff						•									✓
	Water shortage		•						•	•							✓
	Support for low water regimes	Water return to rivers in case of hydric deficit							•	•	•						✓

Annex 6

Adaptive Assessment and Management Approach of the Framework for Freshwater Ecosystem Management: 4 PHASES 9 STEPS



Notes

Freshwater ecosystems such as wetlands, rivers, and lakes are indispensable for life on our planet and vital for directly ensuring a range of benefits and services fundamental to the environment, society and the economy.

However, they face serious pressures which affect their ability to provide those services, such as pollution, over-extraction and encroachment from urban and agricultural development.

One of the main challenges in managing freshwater ecosystems lies in finding the balance between short-term socioeconomic development objectives and the need to protect and restore freshwater ecosystems to support more sustainable, long-term socioeconomic wellbeing.

UN Environment has developed a publication series entitled ‘A Framework for Freshwater Ecosystem Management’. The main aim of the series is to support countries to sustainably manage freshwater ecosystems. In doing so, it supports national and international goals related to freshwater ecosystems, such as certain Aichi Biodiversity Targets and Sustainable Development Goal (SDG) targets. The series currently consists of four volumes:

- Volume 1: Overview and guide for country implementation
- Volume 2: Technical guide for classification and target-setting
- Volume 3: Case studies
- Volume 4: Scientific background for regional consultations on developing water quality guidelines for ecosystems

This volume, ‘Scientific Background”, underpins the series and includes a review of water quality guidelines for ecosystems from around the world. It was the first volume in the series to be developed, beginning in 2013 over a number of years.

UN environment